ELSEVIER

Ecosystem-based management can contribute to cooperation in transboundary fisheries: The case of pacific sardine

Andrés M Cisneros-Montemayor*, Gakushi Ishimura, Gordon R. Munro, U. Rashid Sumaila

The University of British Columbia, Institute for the Oceans and Fisheries, Vancouver, V6T 1Z4, Canada

ABSTRACT

Transboundary fish stocks complicate sustainable fishing strategies, particularly when stakeholders have diverse objectives and regulatory and governance frameworks. Pacific sardine (*Sardinops sagax*) in the California Current is shared by up to three fishing nations— Canada, the United States, and Mexico—and climate-driven abundance and distribution dynamics can complicate cooperative fisheries, leading to overfishing. This study builds on previous analyses by integrating ecosystem linkages into a game theory model of transboundary sardine fisheries under various climate scenarios. Cooperative fishing strategies that account for the ecosystem-wide value of sardine as forage for other species result in increased economic benefits compared to strategies that only account for the single-species value of sardine fisheries to a given fishing country. Total ecosystem landed value is maximized at a sardine fishing rate only somewhat lower than sardine F_{MSY}, which is more precautionary but still allows the fishery to operate. Incorporating ecosystem dynamics into management-applicable models can highlight ways in which ecosystem-based fisheries management can improve both sustainability and profitability and help managers prioritize wider ecological research. Ecosystem-based management will be increasingly required to understand and adapt to the observed rapid shifts in species distributions due to climate change, and to design strategies to achieve sustainable and profitable fisheries amidst changing ecosystems.

1. Introduction

Transboundary stocks add complexity to traditional fisheries management because of a need to account for other users over which they may have no direct influence (Munro, 2009; Munro and Sumaila, 2012). Recent projections of shifting fish distributions due to climate change effects estimate that 23–35% of global EEZs would see new transboundary stocks by 2100 (Pinsky et al., 2018). This study uses a coupled game theory-ecosystem model to explore if ecosystem-based management that accounts for the value of small pelagic species as forage for other fished species can add support for cooperation between countries that fish transboundary stocks. This is particularly important given the known effects of climate on these species, which can rapidly shift their abundance and distribution and therefore complicate cooperative management agreements.

There have been previous frameworks developed to estimate net benefits derived indirectly from a fish stock given ecosystem trophic dynamics and/or game-theoretic fishing strategies. These include, for example, game-theoretical analyses with two linked species and two players (Sumaila, 1997) or one species and three players (Hannesson, 2007; Ishimura et al., 2013a). The comparison between landed value

and supporting service value has also been addressed, either focusing on one species (Herrick Jr, 2010) or a set of species (usually forage fish) within an ecosystem (Cisneros-Montemayor, 2013; Pikitch et al., 2012; Okey et al., 2014). To our knowledge, however, this has not yet been extended into game-theoretic models of transboundary cooperation within an ecosystem framework.

The case of Pacific sardine (*Sardinops sagax*) in the California Current is a useful analytical study species, as it is shared by three major fishing countries, Canada, the United States (US) and Mexico, and is subject to climate-related dynamics that are known but difficult to anticipate and can counteract fishing strategy goals. Game theory models incorporating environmental effects on stock dynamics have indeed shown that this variability can prevent stable cooperative exploitation (Ishimura et al., 2013b). This study moves beyond single-species analyses and incorporates ecosystem linkages. The economic contribution of a species to other fisheries, in the form of supporting services, could shift the balance of the game. Incorporating these ecosystem economic values is vital for developing optimal fishing strategies, particularly given increasing mandates for ecosystem approaches to management (Daily et al., 2009). In this context, ecosystem models are used as a platform to account for benefits to other fisheries

---

following from single-stock management strategies.

The application of game theory in fisheries has provided insightful predictions on the stability of cooperation in transboundary stocks, as well as strategies to improve cooperation when it would not otherwise occur (Munro, 1979). Most advances in game theory applied to cooperative fisheries management have been in the context of increased complexity of the game structure and player strategies (Bailey et al., 2010; Hannesson, 2011b). Cooperative stability can thus be tested given various system shocks, such as changing prices, discount rates, open-access and coalition scenarios (Bjørndal et al., 2000; Pintassilgo and Lindroos, 2006), and environmentally-driven fluctuation of fish stocks (Sumaila, 1999; Bailey et al., 2013; Ishimura et al., 2013a). This continued broadening of game theory from the theoretical to the applied may prove crucial for aiding cooperative management of the world's shared fish stocks (Miller and Munro, 2004), and has already provided a platform for hypothesis testing at different scales (Bailey et al., 2013).

In this context, ecosystem models provide a virtual arena to explore inter-species interactions with all other factors held constant or controlled for. They are used in that capacity here, but it must be stressed that this is not an ecosystem modeling exercise *per se*. Rather, this study uses existing ecosystem models as a tool to test the effects of contributions from a given species to the biomass (and subsequent landed value) of other species in the ecosystem. The results of a range of cooperative and non-cooperative fishing strategies for transboundary Pacific sardine stocks over three countries are then tested to identify the policies that perform best under a range of potential climate scenarios, highlighting the differences in potential outcomes under ecosystem-based versus single-species management.

2. Methods

This study builds on previous game theory models that include temperature-driven population dynamics for a single species (sardine in this case) (Ishimura, Herrick Jr., and Sumaila 2013b; 2013a). The resulting stock distribution along the transboundary range is clearly linked to sea surface temperature (SST) driven by climate and, after fishing rate policies are acted on by each party, is used as an input to other species in a set of three ecosystem models. The results of fishing policy for each party are evaluated in terms of single and multiple species values, over a range of temperature and game-theoretic fishery scenarios. The methods are first presented in general terms so they can be applied to other settings, followed by a detailed description of parameter values and assumptions used for our case study, Pacific sardine and associated ecosystems along the California Current.

2.1. Single-species population dynamics

Following from previous studies (Hannesson, 2007; Ishimura et al., 2013b), temperature (*T*) is assumed to be the ultimate driver of the model, representing prevailing climate regimes. Other climatic indices can easily be employed instead of temperature, but SST has proven to be a very useful indicator for highly variable forage fish populations. Our scenarios include multiple possible time series including both increases and decreases in mean SST over time. In each scenario, the temperature at each time step (*t + 1*), in this case each year, depends on the previous year's temperature (*t*), a trend parameter (*μ*) that indicates if this is a cooling (-*μ*) or warming (+*μ*) scenario, and a uniformly distributed random error term (*σ*, [-1, 1]) to add stochastic variation (Eq. 1) (Ishimura et al., 2013b).

$$T_{t+1} = T_t + \mu \cdot \sigma \tag{1}$$

Because climate can affect not only the abundance, but also the distribution of fish stocks (Cheung et al., 2015), stakeholders along the distribution range can expect varying shares of total abundance. The distribution of the stock ($D_{x,t}$) is calculated as the proportion of the yearly biomass in each player's area, where *x* is one of three players and *t* is the time step, and is estimated as:

$$\begin{cases} for\ x = 1,\ D_{x,t} = max\ \{0,\ min\ [1,\ (T_{H,x} - T_t)/\ T_{H-L,x}\ ]\ \} \\ for\ x = 2,\ D_{x,t} = (1 - D_{1,t}) \cdot max\ \{0,\ min\ [1,\ (T_{H,x} - T_t)/\ T_{H-L,x}\ ]\ \} \\ for\ x = 3,\ D_{x,t} = 1 - (D_{1,t} + D_{2,t}) \end{cases}$$

$$\tag{2}$$

where T_H and $_L$ are the upper and lower temperature thresholds, respectively, for sardine in each country. This equation was developed by Ishimura et al. (Ishimura et al., 2013b), and could be further generalized to allow for more than three players. For transboundary stocks, the share of the stock for any one player at a given time is between 0 and 1, and the sum over all players must always equal 1. For straddling stocks where a portion enters the high seas and might not be accessed by any of the players, this portion could be allocated to an extra "player" representing the inaccessible area.

At each time step, temperature is assumed to have a linear effect on sardine abundance (*B*) through environmental carrying capacity (*K*) (Eq. 3). Biomass and catch are subsequently estimated based on the initial abundance (B_O), distribution (*D*), and fishing rate (*F*) by country:

$$K_t =\ B_0 *\ T_t\ /\ T_{t=1} \tag{3}$$

$$B_{t+1} =\ B_t + r \cdot B_t \cdot (1 - B_t\ /\ K_t\ )\ -\ C_t \tag{4}$$

$$C_t =\ \sum_x\ B_t \cdot D_{x,t} \cdot F_{x,t} \tag{5}$$

Note that, while we assume a positive relationship between temperature and sardine abundance, this assumption can be changed, reversed, or substituted for another functional shape or driver by modifying Eq. 3. Likewise, the rate of population growth (*r*) or the form of the population growth function (Eq. 4) can be substituted for other types that better approximate known dynamics for a given species. In Eq. 5, catch of the transboundary stock (*C*) is assumed to occur at the end of each time step, where *F* is the country-specific fishing rate (see *Game structures* below).

2.2. Ecosystem-level fishery effects

The section above details single-species population dynamics, and can be used to test the payoffs of alternative cooperative strategies in terms of, for example, the landed value of the model organism (e.g. sardine) (Ishimura et al., 2013b). However, ecosystem-based management should consider benefits derived from multiple species with direct and indirect interactions within an ecosystem. Particularly relevant here, the role of forage fishes such as sardines, anchovies and herrings as an energy pathway from planktonic to larger predator species has been highlighted as a key ecosystem supporting service (Bakun et al., 2009; Pikitch et al., 2014), and is captured here.

Ecopath with Ecosim (EwE) is an ecosystem modelling platform that uses mass balance equations and specified trophic linkages to represent an ecosystem (Christensen and Walters, 2004). As with any ecosystem modelling approach, EwE requires more data than a single-species assessment. It has also been limited in terms of its capability to represent spatial dynamics and uncertainty (Plagányi, 2007), although these limitations have increasingly been addressed through an improved spatial component (Ecospace) and the development of guidelines and diagnostics to ensure model coherence (Villasante et al., 2016). EwE is an ideal platform for analyses that aim to contribute to management advice (Plagányi and Butterworth, 2004), such as this study. As the most widely used ecosystem modelling platform, there are hundreds of EwE models available that could be used in different locations using the framework provided here (Palomares et al., 2009) and can be searched through a global repository, EcoBase (Colléter et al., 2015). When more data is available, EwE models can add complexity and include explicit spatial differences in habitat and oceanographic conditions, which has

been used to represent small pelagic fish populations in the Western Pacific (Watari et al., 2019).

In Ecopath, the initial EwE module, the essential parameters for each functional group are biomass (B), production to biomass ratio (P/B), consumption to biomass ratio (Q/B) and ecotrophic efficiency (fraction of biomass used in the system; EE), and a diet matrix specifying the proportion of each functional group in the diet of others. Given these parameters, the production (P) of each functional group (i) is calculated as:

$$P_i = Y_i + \sum_j M_i \cdot B_j + E_i + A_i + O_i \qquad (6)$$

where Y is the total fishery catch rate, M is the instantaneous predation rate, E is the net emigration rate (emigration – immigration), A is the biomass accumulation rate, O is sources of mortality other than fishing and predation and j are predator species (Christensen et al., 2005). Dynamic biomass growth B in time interval t is calculated in the Ecosim module of EwE as:

$$\delta B_i / \delta t = g_i \cdot \sum Q_{ij} - \sum Q_{ji} + I_i - (O_i + F_i + e_i) \cdot B_i \qquad (7)$$

where, for each prey group i and predator group j, g is the net growth efficiency, F is the fishing mortality rate, and e and I are the emigration and immigration rates (Christensen et al., 2005). Consumption rates (Q) are calculated based on the foraging arena concept (Walters et al. 1997), where only a portion of prey biomass (determined by its specific vulnerability parameter) is susceptible to predation at a given time.

Using the built-in biomass forcing tool in Ecosim (EwE version 6.4.2; www.ecopath.org), the relative abundance of a single species (in this case sardine, see *Case study* below) was varied from 0 to 10 times the baseline value, and the equilibrium biomasses for all other species groups were noted for each scenario. The resulting data represent the abundance of each species group given the abundance of sardine (or whatever group is being varied) in the system and allows for a calculation of net impacts on each species from various scenarios of sardine abundance within the game theory component of the model. This establishes the estimated magnitude and direction of changes in species abundance following from changes in sardine abundance, and the actual variation of sardine abundance is each warming or cooling scenario is then driven by SST. Unlike approaches that gauge this contribution using only Ecopath data (Hannesson et al., 2009; Cisneros-Montemayor, 2013), using Ecosim scenarios captures both bottom-up and top-down dynamics (Shannon, 2000).

In Eq. 8, the biomass shares ($B_{x,t}$) of the transboundary stock are estimated at each time step based on the total biomass of the stock and the proportion of the stock for each player (Eq. 2 and 4):

$$B_{x,t} = B_t \cdot D_x \qquad (8)$$

Using a corresponding ecosystem (or multispecies) model as explained above, the relative abundance (A [0,]) of each species group i in country x is calculated as:

$$A_{i,x,t} = \alpha_1 \cdot B_{x,t}^2 + \alpha_2 \cdot B_{x,t} + \alpha_3 + \varepsilon \qquad (9)$$

where the α terms are calculated from a second-order polynomial regression after varying the stock's biomass from 0 to n times the baseline (here, $n = 10$). Combinations of negative and positive α_1 and α_2 values of various magnitudes can represent linear, concave or exponential functions, and α_3 represents the value at the origin, i.e. the abundance of a species when the stock being varied equals zero. Note that, while this study focuses on a forage fish, Eq. 9 can be applied to any species in an ecosystem model and would yield similar results to a keystone analysis comparing the overall impact of individual species' abundance changes (Libralato et al., 2005).

Following from Eq. 9, and assuming that fishing effort remains constant, the yearly landed value (LV) for each species group and country is estimated based on the baseline landed value for each

Table 1
Annual landed value by country off the California Current (CONAPESCA, 2012; DFO, 2013; NMFS, 2014). Ex-vessel price estimated using total catch (metric tons) and value (2014 USD). Average profit margin calculated based on per-gear total fishing costs reported in Lam et al. (Lam et al., 2011). For full table, see Supplementary Files.

| Group | Landed Value (USD millions) | | | | Avg. Price | Avg. Profit |
	Canada	USA	Mexico	Total	(USD 1000/mt)	(%)
Crustaceans	22	240	27	289	4.7	58
Mollusks	20	133	11	164	11.0	87
Benthopelagic Fish	43	48	0.2	91	2.7	14
Salmon	39	49	–	88	4.4	78
Squids	–	66	2	68	0.7	17
Tunas and Billfish	5	50	3	58	3.2	31
Flatfishes	32	21	1	54	2.1	24
Cod-likes	16	22	–	39	0.3	7
Other Invertebrates	6	16	13	35	2.7	57
Sardine	–	22	11	33	0.1	0
Small Pelagic Fish	11	3	0.2	13	0.6	12
Pelagic Fish	3	1	3	8	0.5	32
Other	–	–	5	5	0.6	62
Elasmobranchs	1	1	3	5	0.7	10
All groups	**197**	**673**	**80**	**950**	**2.8**	**30**

country as:

$$LV_{i,x,t} = A_{i,x,t} \cdot LV_{i,x,t=0} \qquad (10)$$

$$NPV_x = \sum_{i,t} (LV_{i,x,t} \cdot \pi_{i,x}) / (1 + d)^t \qquad (11)$$

In Eq. 11, net present value (NPV) from fisheries in each country is estimated assuming a discount rate (d) and profit margins (π) for each species group and player (e.g., Table 1).

The inclusion of available cost data allows for comparison between net present value, total discounted value (the sum of discounted landed value over the time period) and average landed value. The latter is perhaps the most intuitive indicator and is usually much better understood by commercial fishers. These performance metrics are evaluated for each temperature scenario, fishing strategy and country. Note that the effects of different discount rates can have significant implications for policy recommendations, and have been found to be a key potential driver of fishing strategy (Sumaila, 2004; Sumaila and Walters, 2005). This study assumes a discount rate d of 3%, based on the 10-year US government bond rate (U.S. Department of the Treasury, 2019), though this can be easily changed, possibly adopting different discount rates for each country; this was not done here to focus on results due to single- versus ecosystem-wide valuation in the model.

2.3. Game structures

Cooperative coalition structures can be of three types, full (when all players cooperate), partial (when there is at least one non-cooperating free-rider) and non-cooperative (Lindroos et al., 2005). For a three-player game, this results in three possible coalition structures: a grand coalition {x_1, x_2, x_3}, partial cooperation {x_1, x_2} {x_3}, {x_1, x_3} {x_2}, {x_2, x_3} {x_1} and non-cooperation {x_1} {x_2} {x_3}.

This study adapts fishing strategies proposed by Ishimura et al. (Ishimura et al., 2013a) as follows:

- Full cooperation: all players fish at F = fishing rate at maximum sustainable yield (F_{MSY});
- Partial cooperation: cooperating players fish at F = F_{MSY}; free rider fishes at F = 1 if its stock share < 0.5 and F_{MSY} if its stock share > = 0.5;
- Non-cooperation, pragmatic: each player fishes at F = 1 if its stock share < 0.5 and F_{MSY} if its stock share > = 0.5;

- Non-cooperation, windfall: each player fishes at F = 1 if its stock share > 0.5 and F_{MSY} if its stock share < = 0.5;
- No sardine fishing: F = 0 for all players.

Note that the non-cooperative structures seem similar but represent quite different fishing strategies. In the "pragmatic" strategy, players are assumed to fish at a the maximum sustainable yield rate (F_{MSY}) only when they have a large enough stock share in their waters and therefore an incentive to conserve the stock for the future (Hannesson, 2007). If the stock share in their waters is small, players are assumed to instead opt to "fish whatever we can." The "windfall" strategy uses the same reference point, but players opt to fish less conservatively if they are confronted with a small stock share, fishing all they can when natural fluctuations increase abundance in their waters.

In all cases, it is assumed that only annual landed value, total discounted value, and net present value (*NPV*) are evaluated by players, for both single and multiple species. The use of these three metrics is intended to represent the various viewpoints often expressed by resource stakeholders, whose interests may include how much revenue is being made annually, or how much total revenue, or net value (revenue minus costs), can be expected over a given time period. Other metrics, such as minimum allowed population thresholds, are not implemented here, but have been explored in the past in similar modelling exercises as trade-offs to pure market performance (Ishimura et al., 2013a).

2.4. Case study: Pacific sardine in the California Current

The California Current is a Large Marine Ecosystem spanning from the southern coast of British Columbia, Canada, the US coast along Washington, Oregon and California, to the Baja California Peninsula in Mexico (Lynn and Simpson, 1987). A transboundary stock of Pacific sardine is distributed along this ecosystem, and has been previously identified as an useful model organism due to its importance for ecosystems and fisheries, and its large climate and fishery-driven fluctuations in both distribution and abundance (Chavez, 2003; Hannesson and Herrick Jr., 2010).

There are many excellent reviews detailing various aspects of sardine research in the region, including its historical trends (Baumgartner et al., 1992), fishery dynamics (Wolf, 1992; Radovich, 1982), ecological importance (Hannesson et al., 2009; Hannesson and Herrick Jr., 2010) and significance for international cooperative management (Ishimura et al., 2013a, 2013b). Following from Ishimura et al. (Ishimura et al., 2013a, 2013b) to compare results with prior single-species game theory models, this study assumes an initial biomass (B_0) of 1.2 million tonnes of sardine along the California Current (estimated around the year 2000), and a population growth rate (*r*) of 0.27. These parameter and population dynamics assumptions can be easily modified to better describe a model species.

Using the framework detailed above, the ecosystem-wide economic impact to each player (Mexico, US, and Canada) of regional sardine abundance as it relates to climate change is estimated and used as an alternative objective function to the single-species approach developed in Ishimura et al. (Ishimura et al., 2013a, 2013b). This is performed here using two separate ecosystem models for the California Current representing Canadian and US areas.

The Canada model was developed to represent the southwest coast of Vancouver Island, British Columbia, the main area where Pacific sardine is distributed in the region. Parameters were based on published models developed for the Strait of Georgia (Martell et al., 2002), southern BC shelf (Pauly and Christensen, 1996), BC shelf (Preikshot, 2005) and northern California Current (Field et al., 2006). Input parameters and trophic linkages are presented in the Supplementary Files. Of the 33 groups represented in the model, 19 were linked directly to sardine either as predators or prey. The US model (Field et al., 2006; Field and Francis, 2006) includes 25 species groups, though six are further represented by multiple life stages (i.e., larval, juvenile,

adult) (Supplementary Files). In this model, only four groups are directly linked to sardine. There were no available models for this area of Mexico, so the US model is used as a proxy. This is not ideal and assumes that fisheries have had comparable impacts on populations and ecosystems on both sides of the border. Without a current alternative, we used this model under the assumption that the marine ecosystems of northern Baja California and adjacent California—that are indeed part of the same California Current Large Marine Ecosystem—have enough similarity to be able to use models for the latter in this conceptual study. For example, the US model was itself built to test similar climate change hypotheses and includes Pacific sardine as a distinct group (Field et al., 2006). However, we caution that results should be considered in terms of the relative differences between alternative fishing strategies, and that absolute USD values are included only to better convey the magnitude of changes stemming from our scenarios.

Available data on current catch and landed values for the three countries along the California Current was gathered from official sources (CONAPESCA, 2012; DFO, 2013; NMFS, 2014) and aggregated by broader species groups to allow for better comparisons. According to the most recent available official data, the California Current supports fisheries with a total landed value of approximately US$950 million per year (Table 1). The same species groupings were assigned to species in the ecosystem models after calculating the effects of changes in sardine abundance (polynomial regression given 0 to 10 times baseline sardine abundance) to obtain the α parameters in Eq. 9.

Profit margins (π in Eq. 11) were calculated based on total (including fixed and variable) fishing costs per tonne for corresponding fishing gears and players (Lam et al., 2011) (Table 1, Supplementary File). Based on these cost estimates and official price data (CONAPESCA, 2012; DFO, 2013; NMFS, 2014), in some cases the reported fishing costs per tonne exceeded the reported ex-vessel price for the same species; profit margin was set to zero in these cases. This result may be partly due to the resolution of cost data (Lam et al., 2011), but could also reflect the effect of profit-enhancing subsidies (Sumaila et al., 2010) or vertically-integrated fishing firms operating their fishing sector at a loss that is made up in the processing sector (Sumaila et al., 2012). This calculation uses available data to provide an approximation to fishing cost, but more importantly does not include bioeconomic dynamics, specifically effort-dependent cost and abundance-dependent catch per unit effort. This is one key area for improvement in future work.

These parameters, together with baseline landed values and fishing strategy effects on sardine were used to estimate subsequent landed values and net present value by country and species (Eq. 10, 11). Fishing strategies were implemented as detailed above, with two partial cooperation scenarios where either Mexico or Canada are free riders. Biomass of other species given sardine abundance represent equilibrium model results.

3. Results

A representative net impact on species (aggregated into species groups) over all countries is presented in Fig. 1, showing Ecosim-modelled relative biomass for species groups after a ten-fold increase in relative sardine abundance compared to baseline.

All other things being equal, average sardine catch (and value) increases as fishing rate on sardine is increased from zero to a maximum at F_{MSY} (Fig. 2). The concurrent reduction in sardine biomass negatively impacts catch and value for other fisheries due to a net reduction in available prey (Fig. 1). This loss is offset by sardine landed value so that the total value is increased with sardine F up until just less than the sardine F_{MSY} (Fig. 2). Under current assumptions, a single owner would thus maximize total sardine annual landed value by fishing at F = 0.17, total annual landed value at F = 0.15, and annual landed value of all species other than sardine at F = 0 (Fig. 2). At fishing rates higher than sardine F_{MSY}, however, reductions in sardine abundance due to
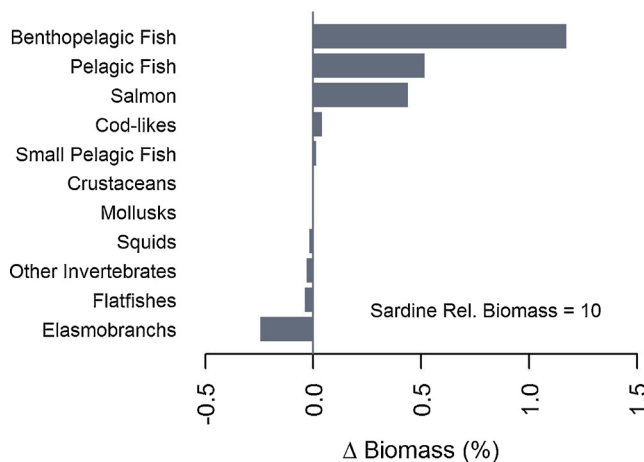
Fig. 1. Net biomass change (including all countries) for fished groups in the California Current ecosystem models given change to Pacific sardine abundance.
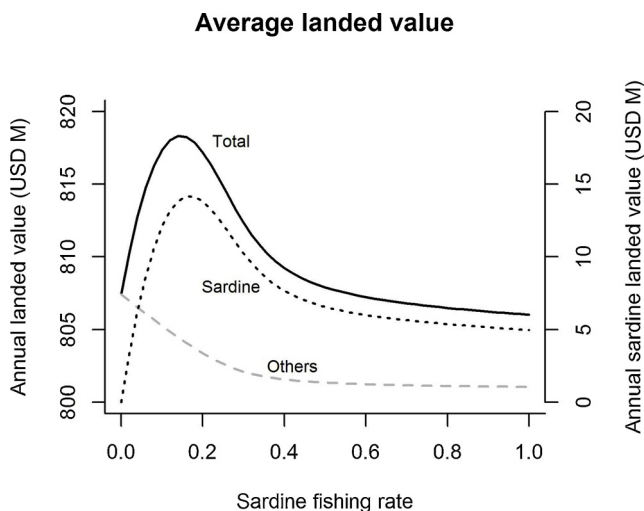

Fig. 2. Total and sardine average landed value per year at baseline scenario given sardine fishing rate applied to entire stock, all other things being equal. Sardine annual landed value maximized at F = 0.17; Total annual landed value maximized at F = 0.15.

overfishing result in a net decrease in sardine, other species', and total landed value (Fig. 2). Note that only species that appear both in official catch statistics and ecosystem models are included, so that landed values in results are slightly less than the total from available fisheries data (Table 1).

Each of the various game structures can have different performance given prevailing temperature trends, both for sardine and total landed values (Fig. 3). Among all strategies, cooperation between at least two players outperforms other management structures (Fig. 3).

In Table 2, results are presented relative to the performance of the non-cooperative pragmatic fishing strategy ('Baseline'), the most similar strategy to current transboundary management, over all temperature scenarios. This provides a point of reference for outcomes of moving away from current management. Table 3 presents results relative to the average for all strategies, and represents the relative value, over all temperature scenarios, of selecting a particular strategy compared to choosing randomly among all strategies.

Results by player show similar trends but vary in magnitude between the three countries, Mexico (Table 4), Canada (Table 5) and the US (Table 6). The outcomes in these tables are again shown relative to the mean over all strategies and temperature scenarios. Mexico

(Table 3) gains the most from free-riding compared to the other players, who see greater losses to NPV from non-cooperation (Table 4, 5). Climate scenarios modify each player's optimal strategy, where Mexico or Canada benefit the most from non-cooperation when sardine distribution shifts into their water in the warming and cooling scenarios, respectively (Table 3, 4). Both of these strategies lower U.S. (Table 5) and overall payoffs, yet full cooperation results in the best payoffs in terms of sardine and total discounted value.

4. Discussion

This study explored the potential implications for transboundary management of fully incorporating an ecosystem-based approach. Results from the ecosystem-based game theory model suggest that cooperative fishing strategies outperform others over a range of temperature scenarios when goals incorporate ecosystem-wide economic value. Fisheries along large marine ecosystems like the California Current rarely focus on a single species (e.g., Table 1), and therefore management should incorporate species interactions (at the very least) into policies for sustainable fishing. The influence of forage fishes on overall production (e.g., Fig. 1) and subsequent fisheries catch has been identified for many marine ecosystems around the world (Shannon, 2000; Hannesson and Herrick Jr., 2010; Pikitch et al., 2012; Okey et al., 2014). Managers should be aware of the key role played by small pelagic fishes and other lower-trophic level species and likely aim for fishing mortality rates that are lower than MSY but overall positive for other fisheries and the wider ecosystem (Fig. 2; Smith et al., 2011).

As the scope and scale of challenges for fisheries management—including climate change effects, gaps in legal frameworks, market and subsidy dynamics, etc.—become increasingly clear, there has been a concurrent increase in efforts to promote international agreements to achieve and maintain sustainable fisheries. These include the ongoing discussions on management in areas beyond national jurisdiction (Blasiak et al., 2016), curtailing of harmful fisheries subsidies (World Trade Organization, 2017), and the explicit inclusion of ocean issues in the UN Sustainable Development Goals (UN, 2015). The optimal management of transboundary fish stocks has been a similar concern historically in some ocean regions including the North Pacific and North Atlantic, but the projected increase in transboundary stocks linked to climate change effects will require more such arrangements, including in contexts where data availability and management capacity are uneven among nations.

While Pacific sardine have supported important stocks at different times along the California Current, management objectives can be quite different for the countries along this LME, or within countries. For example, sardine catch in Mexico is overwhelmingly destined for reduction into fish meal, but they are also used as feed for tuna ranching on the northern coast of Baja California, and some segments of the fishery are certified by the Marine Stewardship Council (MSC, 2019). This study highlights the high complexity of transboundary management, yet only three players, six game strategies, and seven metrics (annual and total discounted value of sardine, others and total, and NPV) are evaluated. Transboundary, straddling or high-seas stocks often involve many more players (Munro, 1979; Sumaila et al., 2015; Hannesson, 2011a) and strategies are very rarely held over time as they have been modelled here. This is made significantly more complicated by the inclusion of ecosystem-level complexity, which could be expanded to account for the objectives of targeted fisheries on specific species, and even non-market values (Daily et al., 2009). When analyzing results with a view to modifying policy, players should consider not just the estimated performance of the best strategies (Table 3), but also their benefits relative to current strategies (Table 2) and the potential costs (e.g., transactional and institutional) of moving away from them. Incorporating supporting service values—such as the role of sardine and other small pelagics as forage—can indeed change optimal policy as shown by our results, but this can also follow from wider valuations
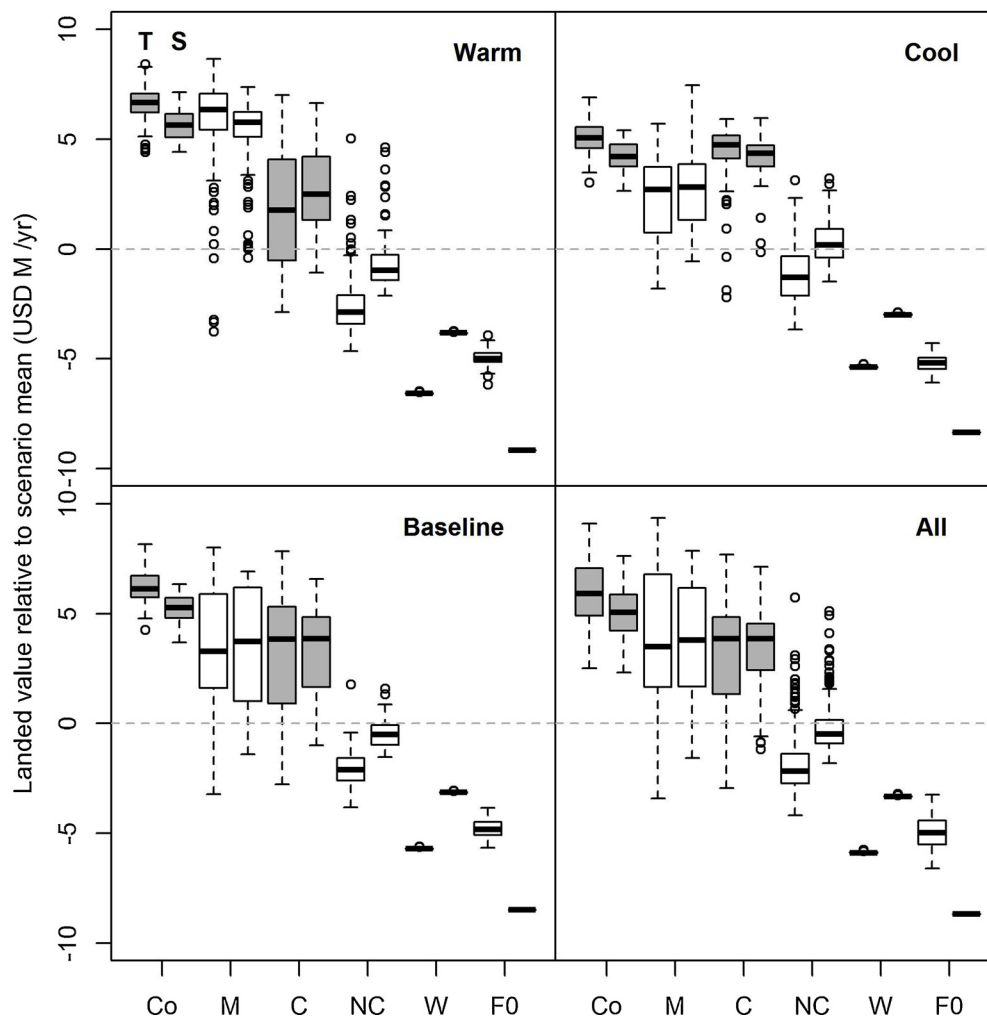
Fig. 3. Total and sardine annual landed value per year given multi-player fishery strategy, relative to the overall mean for each temperature scenario (n = 10,000). Co = full cooperation; M = Mexico free rider; C = Canada free rider; NC = non-cooperative pragmatic (players fish all available sardine when their stock share < 0.5), W = non-cooperative windfall (players fish all available sardine when their stock share > 0.5); F0= no sardine fishing.

accounting for fishing costs, which may be a more cost-effective prior analysis to undertake.

The analyses using ecosystem models representing the California Current lend support for a net positive effect of sardine abundance (acting as an independent variable) on the abundance of other species and their subsequent landed values. If this effect were quite pronounced, the optimal solution would be to leave sardine unfished so that other fisheries would benefit, thereby increasing total value (Sumaila, 1997). However, the magnitude of the relationships found in

this case are not enough to offset losses from foregone sardine catch (F0 in Fig. 3) so that annual total landed value is maximized at a fishing rate above zero, just below F_{MSY} (Fig. 2, Table 2, 3). A very similar result has been found in previous research examining the ecosystem effects of fisheries for low trophic level species (including small pelagic fishes), that suggest ecosystem impacts can be significantly reduced by fishing forage species at around 80% of MSY (Smith et al., 2011).

Another key source of uncertainty is that, though this is not intended as an ecosystem modelling exercise *per se*, results evidently

Table 2
Performance of management strategies for all players relative to non-cooperative (pragmatic) strategy, over all climate simulations (n = 1000). Values represent mean deviations from the baseline, in 2014 USD millions.

Total relative to non-cooperative	Annual landed value (USD millions /year)			Total discounted value (USD millions)			Net present Value
Strategy	Sardine	Others	Total	Sardine	Others	Total	(USD millions)
Full Cooperation	5.3	2.7	8	65	59	125	14.9
Mexico Free Rider	4.2	1.6	5.8	64	33	97	8.6
Canada Free Rider	3.7	1.5	5.1	53	31	84	8
Non-Cooperative (Windfall)	−3	−0.7	−3.7	−66	−19	−86	−3.9
No Sardine Fishing	−8.4	5.6	−2.8	−257	119	−138	30.3
Baseline (Non-cooperative pragmatic)	**8**	**802**	**810**	**257**	**18,037**	**18,294**	**7,980**

Table 3

Performance of cooperative management strategies for all players relative to mean over all strategies and climate simulations (n = 1000). Values represent mean deviations from the baseline, in 2014 USD millions.

Total relative to all strategies	Annual landed value (USD millions /year)			Total discounted value (USD millions)			Net present Value
Strategy	Sardine	Others	Total	Sardine	Others	Total	(USD millions)
Full Cooperation	5	1	6	89	22	111	5.3
Mexico Free Rider	3.9	− 0.2	3.7	87	− 4	83	− 1
Canada Free Rider	3.4	− 0.3	3.1	77	− 6	71	− 1.7
Non-Cooperative (Pragmatic)	− 0.3	− 1.8	− 2.1	23	− 37	− 13	− 9.6
Non-Cooperative (Windfall)	− 3.3	− 2.5	− 5.8	− 43	− 56	− 100	− 13.6
No Sardine Fishing	− 8.7	3.8	− 4.9	− 233	82	− 152	20.7
Baseline (All strategies)	**8.7**	**803**	**812**	**− 233**	**18,074**	**18,308**	**7,982**

Table 4

Performance of management strategies for Mexico relative to mean over all strategies and climate simulations (n = 1000). Values represent mean deviations from the baseline, in 2014 USD millions.

Mexico relative to all strategies	Annual landed value (USD millions /year)			Total discounted value (USD millions)			Net present Value
Strategy	Sardine	Others	Total	Sardine	Others	Total	(USD millions)
Full Cooperation	1.1	0	1.1	13	0.1	13	0.1
Mexico Free Rider	2.5	0	2.5	72	0	72	0
Canada Free Rider	0.7	0	0.7	5	0	5	0
Non-Cooperative (Pragmatic)	0.6	− 0.01	0.6	31	− 0.1	31	− 0.1
Non-Cooperative (Windfall)	− 2.3	− 0.01	− 2.4	− 58	− 0.2	− 58	− 0.2
No Sardine Fishing	− 2.5	0.01	− 2.5	− 63	0.2	− 63	0.3
Baseline (All strategies)	**2.5**	**57**	**59.5**	**63**	**1,282**	**1,345**	**1,042**

Table 5

Performance of cooperative management strategies for Canada relative to mean over all strategies and climate simulations (n = 1000). Values represent mean deviations from the baseline, in 2014 USD millions.

Canada relative to all strategies	Annual landed value (USD millions /year)			Total discounted value (USD millions)			Net present Value
Strategy	Sardine	Others	Total	Sardine	Others	Total	(USD millions)
Full Cooperation	1.4	0.8	2.2	19	18	38	3.1
Mexico Free Rider	0.9	− 0.2	0.7	9	− 3.7	5.4	− 0.6
Canada Free Rider	2.4	− 0.3	2.1	68	− 5.2	63	− 1
Non-Cooperative (Pragmatic)	0.2	− 1.5	− 1.3	21	− 32	− 11	− 5.6
Non-Cooperative (Windfall)	− 2.4	− 2.2	− 4.6	− 57	− 49	− 106	− 7.8
No Sardine Fishing	− 2.5	3.3	0.8	− 60	− 71	10	12
Baseline (All strategies)	**2.5**	**147**	**150**	**60**	**3,311**	**3,371**	**868**

hinge on the ecosystem models used and their assumed trophic relationships and strengths. It is quite possible that other model configurations could change the magnitude of results though perhaps not change trends—that would imply very different ecosystem structure than current knowledge suggests. It is particularly important to encourage the development of ecosystem models in the Mexican California Current given the importance of fisheries for this region and the high existing scientific knowledge and capacity in the country (Palacios-Abrantes et al., 2019). Furthermore, because the regressions (Eq. 9) denote equilibrium biomass, our method assumes that species will respond to sardine abundance instantaneously each year, which clearly may not be the case given large and sudden fluctuations. More importantly, the lack of explicit incorporation of non-trophic

relationships between species is an issue that should be addressed in future applications. Sardine in particular has been known to have correlations (causal or otherwise) with abundances of other small pelagics in the California Current (Baumgartner et al., 1992; Chavez, 2003). Similar relationships have been observed in other pelagic systems (e.g., the Benguela Current; (Shannon et al., 2004)), and it is likely that many more have not yet been identified.

This conclusion holds for total landed value but incorporating costs of fishing can change perceptions on optimal strategy. The net present value by country was calculated for all species based on available data on fishing costs per tonne, which reports a zero per cent profit margin for sardine fisheries (Lam et al., 2011). This plays a large part in the result that, once fishing costs have been accounted for, the net present

Table 6

Performance of cooperative management strategies for the U.S. relative to mean over all strategies and climate simulations (n = 1000). Values represent mean deviations from the baseline, in 2014 USD millions.

U.S. relative to all strategies	Average landed value (USD millions /year)			Total discounted value (USD millions)			Net present Value
Strategy	Sardine	Others	Total	Sardine	Others	Total	(USD millions)
Full Cooperation	2.6	0.1	2.7	56	2.9	59	2.1
Mexico Free Rider	0.4	0	0.4	5.6	−0.5	5.1	−0.4
Canada Free Rider	0.3	0	0.3	3.4	−0.8	2.6	−0.7
Non-Cooperative (Pragmatic)	−1.1	−0.2	−1.3	−28	−4.9	−33	−3.9
Non-Cooperative (Windfall)	1.5	−0.3	1.2	72	−7.5	65	−5.6
No Sardine Fishing	−3.7	0.5	−3.2	−109	10.8	−99	8.4
Baseline (All strategies)	**3.7**	**599**	**603**	**−109**	**13,480**	**13,589**	**6,071**

value for all fisheries is indeed maximized when sardine is left unfished (Table 3). However, the difference between this first-best strategy is much lower in Mexico (Table 4) than in the US (Table 6) or Canada (Table 5), thus modifying each individual player's incentives. For example, in terms of total landed value Mexico gains the most from free-riding, with little effect on NPV (Table 3), compared to the other players, that see greater losses to NPV from non-cooperation (Table 4, 5). Notably, the second-best strategy for any player's NPV remains full cooperation. There are currently no such cooperative agreements for transboundary stocks between the three countries, though a Trinational Sardine Forum attended by academics, government researchers, and industry representatives convenes yearly to share research and discuss market trends and could be a platform to support more formal cooperative fisheries management. In addition to the benefits of collaboration among scientists, fishers, and managers, it is essential that the contexts and objectives for fisheries in different countries (or regions, or fleets) are clear to other users so that they can engage in more productive discussions on cooperation.

Cooperative strategies even with at least two players consistently outperform other structures (Fig. 3), which lends support for potential side payments as a strategy to bring one player into full cooperation (Munro, 1979). Side payments are a form of profit-sharing useful when the total payoffs from cooperative strategies are high, but individual payoffs for a given participant are low. So, a portion of total payoffs are re-distributed to incentivize those participants and facilitate cooperation. This strategy has been employed in a range of settings, from fisheries to water use, and could indeed be applied to the California Current case even with a single-species approach (Ishimura et al., 2013b; Bailey et al., 2013). However, the wider analysis used here contributes to the calculation of required compensation, as a participant could argue that losses in terms of foregone sardine catch (for example if another player were asked to decrease their fishing rate in favor of the cooperative policy) should be tempered by the potential gains for other fisheries, thus reducing the payment amount.

It is increasingly practical to develop models—limitations notwithstanding—that incorporate ecosystem linkages and components that are useful for managers (Trites et al., 1999; Plagányi, 2007; Fulton, 2010; Cisneros-Montemayor et al., 2012). Nevertheless, fully representing or modelling the complexities of real-world social-ecological systems is impossible or impractical. Managers must therefore consider how much more useful further information or a more intricate management system might be in relation to increased research and management costs. These types of questions can be informed by relatively straightforward scenario testing as shown in this study. In the case of the California Current, integrating a bioeconomic and ecosystem approach to management adds to arguments for sustainable single-stock harvest of sardine. Managers must be aware of the key role played by forage species in marine ecosystems, and ecosystem-based management

provides a framework to identify, evaluate, and integrate species linkages in management strategies. This wider and more appropriate view can change the fishing strategies perceived as optimal in transboundary and in multi-stakeholder fisheries, resulting in increased cooperation, and, ultimately, more profitable and sustainable fisheries for all users.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.fishres.2019.105401.

References

Bailey, Megan, Ishimura, Gakushi, Paisley, Richard, Rashid Sumaila, U., 2013. Moving beyond catch in allocation approaches for internationally shared fishstocks. Mar. Policy 40 (July), 124–136. https://doi.org/10.1016/j.marpol.2012.12.014.

Bailey, Megan, Rashid Sumaila, U., Lindroos, Marko, 2010. Application of game theory to fisheries over three decades. Fish. Res. 102 (1), 1–8.

Bakun, Andrew, Babcock, Elizabeth A., Lluch-Cota, Salvador E., Santora, Christine, Salvadeo, Christian J., 2009. Issues of ecosystem-based management of forage fisheries in 'Open' non-stationary ecosystems: the example of the sardine fishery in the Gulf of California. Rev. Fish Biol. Fish. 20 (1), 9–29. https://doi.org/10.1007/s11160-009-9118-1.

Baumgartner, T.E., Soutar, A., Ferreira-Bartrina, V., 1992. Reconstruction of the history of pacific sardine and northern anchovy populations over the past two millennia from sediments of the Santa Barbara Basin, California. Calif. Coop. Ocean. Fish. Inves., Rep 33, 24–40.

Bjørndal, Trond, Kaitala, Veijo, Lindroos, Marko, Munro, Gordon R., 2000. The management of high seas fisheries. Ann. Oper. Res. 94 (1), 183–196.

Blasiak, Robert, Pittman, Jeremy, Yagi, Nobuyuki, Sugino, Hiroaki, 2016. Negotiating the use of biodiversity in marine areas beyond national jurisdiction. Front. Mar. Sci. 3 (November). https://doi.org/10.3389/fmars.2016.00224.

Chavez, F.P., 2003. From anchovies to sardines and back: multidecadal change in the Pacific Ocean. Science 299 (5604), 217–221. https://doi.org/10.1126/science.1075880.

Cheung, William W.L., Brodeur, Richard D., Okey, Thomas A., Pauly, Daniel, 2015. Projecting future changes in distributions of pelagic fish species of northeast pacific shelf seas. Prog. Oceanogr. 130 (January), 19–31. https://doi.org/10.1016/j.pocean.2014.09.003.

Christensen, Villy, Walters, Carl J., 2004. Ecopath with ecosim: methods, capabilities and limitations. Ecol. Modell. 172 (2–4), 109–139. https://doi.org/10.1016/j.ecolmodel.2003.09.003.

Christensen, Villy, Walters, Carl J., Pauly, Daniel, 2005. Ecopath With Ecosim: a User's Guide. Fisheries Centre, University of British Columbia, Vancouver, pp. 154. ftp://ftp.fisheries.ubc.ca/ecopath/webfiles/Ewe%20User%20Guide%205_1.pdf.

Cisneros-Montemayor, A.M., 2013. Modelos Ecológicos Para Analizar El Papel Económico de Los Peces de Forraje. In: Blanco, J.A. (Ed.), Aplicaciones de Modelos Ecológicos a La Gestión de Recursos Naturales. OmniaScience, Barcelona, pp. 67–76.

Cisneros-Montemayor, A.M., Christensen, Villy, Arreguín-Sánchez, Francisco, Rashid Sumaila, U., 2012. Ecosystem models for management advice: an analysis of

recreational and commercial fisheries policies in Baja California Sur, Mexico. Ecol. Modell. 228 (March), 8–16. https://doi.org/10.1016/j.ecolmodel.2011.12.021.

Colléter, Mathieu, Valls, Audrey, Guitton, J.érôme, Gascuel, Didier, Pauly, Daniel, Christensen, Villy, 2015. Global overview of the applications of the ecopath with ecosim modeling approach using the EcoBase models repository. Ecol. Modell. 302 (April), 42–53. https://doi.org/10.1016/j.ecolmodel.2015.01.025.

CONAPESCA, 2012. Anuario Estadístico De Acuacultura Y Pesca. 2012. SAGARPA, México. https://www.gob.mx/conapesca/documentos/anuario-estadistico-de-acuacultura-y-pesca.

Daily, Gretchen C., Polasky, Stephen, Goldstein, Joshua, Kareiva, Peter M., Mooney, Harold A., Pejchar, Liba, Ricketts, Taylor H., Salzman, James, Shallenberger, Robert, 2009. Ecosystem services in decision making: time to deliver. Front. Ecol. Environ. 7 (1), 21–28. https://doi.org/10.1890/080025.

DFO, 2013. Commercial Fisheries Landings, by Province. Department of Fisheries and Oceans. http://www.dfo-mpo.gc.ca/stats/commercial/sea-maritimes-eng.htm.

Field, J.C., Francis, R.C., Aydin, K., 2006. Top-down modeling and bottom-up dynamics: linking a fisheries-based ecosystem model with climate hypotheses in the Northern California current. Prog. Oceanogr. 68 (2–4), 238–270. https://doi.org/10.1016/j.pocean.2006.02.010.

Field, John C., Francis, Robert C., 2006. Considering ecosystem-based fisheries management in the California current. Mar. Policy 30 (5), 552–569. https://doi.org/10.1016/j.marpol.2005.07.004.

Fulton, Elizabeth A., 2010. Approaches to end-to-End ecosystem models. J. Mar. Syst. 81 (1–2), 171–183. https://doi.org/10.1016/j.jmarsys.2009.12.012.

Hannesson, R., Herrick Jr., S.F., 2010. The value of pacific sardine as forage fish. Mar. Policy 34, 935–942.

Hannesson, 2007. Global warming and fish migrations. Nat. Resour. Model. 20, 301–319.

Hannesson, 2011a. Rights based fishing on the high seas: is it possible? Mar. Policy 35 (5), 667–674. https://doi.org/10.1016/j.marpol.2011.02.007.

Hannesson, 2011b. Game theory and fisheries. Annu. Rev. Resour. Economics 3 (1), 181–202. https://doi.org/10.1146/annurev-resource-083110-120107.

Hannesson, Herrick Jr, S.F., Field, John, 2009. Ecological and economic considerations in the conservation and management of the pacific sardine (Sardinops sagax). Can. J. Fish. Aquat. Sci. 66 (5), 859–868. https://doi.org/10.1139/F09-045.

Ishimura, Gakushi, Herrick Jr, S.F., Sumaila, U.R., 2013a. Fishing games under climate variability: transboundary management of Pacific Sardine in the California current system. Environ. Econ. Policy Stud. 15 (2), 189–209. https://doi.org/10.1007/s10018-012-0048-0.

Ishimura, Gakushi, Herrick Jr, S.F., Sumaila, U.R., 2013b. Stability of cooperative management of the pacific sardine fishery under climate variability. Mar. Policy 39, 333–340.

Lam, V.W.Y., Sumaila, U.R., Dyck, A., Pauly, D., Watson, R., 2011. Construction and first applications of a global cost of fishing database. Ices J. Mar. Sci. 68 (9), 1996–2004.

Libralato, S., Christensen, Villy, Pauly, Daniel, 2005. A method for identifying keystone species in food web models. Ecol. Modell. 195 (304), 153–171.

Lindroos, Marko, Kronbak, Lone Grønb\aek, Kaitala, Veijo, 2005. Coalition games in fisheries economics. Advances in Fisheries Economics: Festschrift in Honour of Professor Gordon R. Munro 184–195.

Lynn, Ronald J., Simpson, James J., 1987. The California current system: the seasonal variability of its physical characteristics. J. Geophys. Res. 92 (12) 947–12.

Martell, S.J.D., Beattie, A., Walters, C.J., Nayar, T., Briese, R., 2002. The use of ecosystem models to investigate multispecies management strategies for capture fisheries. In: In: Pitcher, Tony J., Cochrane, K.L. (Eds.), Fisheries Centre Research Reports 2. Fisheries Centre Vol. 10 University of British Columbia.

Miller, Kathleen A., Munro, Gordon R., 2004. Climate and cooperation: a new perspective on the management of shared fish stocks. Mar. Resour. Econ. 19 (3), 367–393.

MSC, 2019. Small Pelagics Fishery in Sonora, Gulf of California 2019. https://fisheries.msc.org/en/fisheries/small-pelagics-fishery-in-sonora-gulf-of-california/.

Munro, Gordon R., 2009. Game theory and the development of resource management policy: the case of international fisheries. Environ. Dev. Econ. 14 (01), 7–27.

Munro, G.R., 1979. The optimal management of transboundary renewable resources. Can. J. Econ. 13 (3), 355–376.

Munro, G.R., Sumaila, U.R., 2012. Subsidies and the Sustainability of Offshore Fisheries: An Economist's Perspective. World Wildlife Fund, In, 15. La Paz, Mexico.

NMFS, 2014. Annual Landings by Species (Washington, Oregon, California). National Marine Fisheries Service. http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings-with-group-subtotals/index.

Okey, T.A., Cisneros-Montemayor, A.M., Pugliese, R., Sumaila, R., 2014. Exploring the trophodynamic signatures of forage species in the US South Atlantic bight ecosystem to maximize system-wide value. 2014–14. Fisheries Centre Working Paper Series. The University of British Columbia.

Palacios-Abrantes, Juliano, Cisneros-Montemayor, Andrés M., Cisneros-Mata, Miguel A., Rodríguez, Laura, Arreguín-Sánchez, Francisco, Aguilar, Verónica, Domínguez-Sánchez, Santiago, et al., 2019. A metadata approach to evaluate the state of ocean knowledge: strengths, limitations, and application to Mexico. Edited by Andrea Belgrano. PLoS One 14 (6), e0216723. https://doi.org/10.1371/journal.pone.0216723.

Palomares, M.L., Morissette, Lyne, Cisneros-Montemayor, A.M., Varkey, Divya, Coll, Marta, Piroddi, Chiara, 2009. Ecopath 25 years conference proceedings: extended abstracts. 17 (3). Fisheries Centre Research Reports. The University of British Columbia. https://circle.ubc.ca/handle/2429/40929.

Pauly, D., Christensen, V., 1996. Mass-balance models of the North-Eastern pacific ecosystem. 4 (1). Fisheries Centre Research Reports. The University of British Columbia, Vancouver.

Pikitch, Ellen K., Boersma, P.D., Boyd, I.L., Conover, D.O., Cury, Philippe, Essington, T.E., Heppell, Selina S., et al., 2012. Little Fish, Big Impact: Managing a Crucial Link in Ocean Food Webs. Lenfest Ocean Program, Washington, D.C.

Pikitch, Ellen K., Rountos, Konstantine J., Essington, Timothy E., Santora, Christine, Pauly, Daniel, Watson, Reg, Sumaila, Ussif R., et al., 2014. The global contribution of forage fish to marine fisheries and ecosystems. Fish Fish. 15 (1), 43–64. https://doi.org/10.1111/faf.12004.

Pinsky, M.L., Reygondeau, Gabriel, Caddell, Richard, Palacios-Abrantes, J.E., Spijkers, Jessica, Cheung, W.W.L., 2018. Preparing ocean governance for species on the move. Science 360 (6394), 1189–1191.

Pintassilgo, P., Lindroos, Marko, 2006. Coalition Formation in High Seas Fisheries: A Partition Function Approach In. Zaragoza, Spain. http://oceandocs.org/handle/1834/854.

Plagányi, É.E., 2007. Models for En Ecosystem Approach to Fisheries." FAO Fisheries Technical Paper 477. FAO, Rome.

Plagányi, E.E., Butterworth, D.S., 2004. A critical look at the potential of Ecopath with ecosim to assist in practical fisheries management. Afr. J. Mar. Sci. 26, 261–287. https://doi.org/10.2989/18142320409504061.

Preikshot, Dave., 2005. Data sources and derivation of parameters for generalised northeast pacific ocean ecopath with ecosim models. UBC Fish. Centre Res. Rep. 13, 179–206.

Radovich, John., 1982. The collapse of the California sardine fishery. What Have We Learned 56–77.

Shannon, L., 2000. Modelling effects of fishing in the Southern Benguela ecosystem. Ices J. Mar. Sci. 57 (3), 720–722. https://doi.org/10.1006/jmsc.2000.0716.

Shannon, Lynne J., Field, John G., Moloney, Coleen L., 2004. Simulating Anchovy–Sardine Regime Shifts in the Southern Benguela Ecosystem. Ecol. Modell. 172 (2–4), 269–281. https://doi.org/10.1016/j.ecolmodel.2003.09.011.

Smith, A.D.M., Brown, C.J., Bulman, C.M., Fulton, E.A., Johnson, P., Kaplan, I.C., Lozano-Montes, H., et al., 2011. Impacts of Fishing Low-Trophic Level Species on Marine Ecosystems. Science 333 (6046), 1147–1150. https://doi.org/10.1126/science.1209395.

Sumaila, U.R., Khan, Ahmed S., Dyck, Andrew J., Watson, Reg, Munro, Gordon, Tydemers, Peter, Pauly, Daniel, 2010. A Bottom-up Re-Estimation of Global Fisheries Subsidies. J. Bioeconomics 12 (3), 201–225. https://doi.org/10.1007/s10818-010-9091-8.

Sumaila, U.R., Lam, Vicky W.Y., Miller, Dana D., Teh, Louise, Watson, Reg A., Zeller, Dirk, Cheung, William W.L., et al., 2015. Winners and losers in a world where the high seas is closed to fishing. Nat. Sci. Rep. 5 (February), 8481. https://doi.org/10.1038/srep08481.

Sumaila, Ussif R., Walters, Carl, 2005. Intergenerational discounting: a new intuitive approach. Ecol. Econ. 52 (2), 135–142. https://doi.org/10.1016/j.ecolecon.2003.11.012.

Sumaila, Ussif Rashid, 1997. Strategic dynamic interaction: the case of Barents Sea Fisheries. Mar. Resour. Econ. 12, 77–94.

Sumaila, Ussif Rashid, 1999. A review of game-theoretic models of fishing. Mar. Policy 23 (1), 1–10.

Sumaila, Ussif Rashid, 2004. Intergenerational cost–Benefit analysis and marine ecosystem restoration. Fish Fish. 5 (4), 329–343.

Sumaila, Ussif Rashid, Cheung, William, Dyck, Andrew, Gueye, Kamal, Huang, Ling, Lam, Vicky, Pauly, Daniel, et al., 2012. Benefits of rebuilding global marine fisheries outweigh costs." edited by julian clifton. PLoS One 7 (7), e40542. https://doi.org/10.1371/journal.pone.0040542.

Trites, A.W., Livingston, P.A., Vasconcellos, M.C., Mackinson, S., Springer, A.M., Pauly, Daniel, 1999. Ecosystem Considerations and the Limitations of Ecosystem Models in Fisheries Management: Insights From the Bering Sea. AK-SG-99-01. Alaska Sea Grant College Program.

UN, 2015. Transforming Our World: The 2030 Agenda for Sustainable Development. A/Res/70/1. UN General Assembly. United Nations, New York. https://www.ceeol.com/search/article-detail?id=304581.

U.S. Department of the Treasury, 2019. Daily Treasury Yield Curve Rates. https://www.treasury.gov/resource-center/data-chart-center/interest-rates/pages/TextView.aspx?data=yieldYear&year=2018.

Villasante, S., Arreguín-Sánchez, F., Heymans, J.J., Libralato, S., Piroddi, C., Christensen, V., Coll, M., 2016. Modelling marine ecosystems using the ecopath with ecosim food web approach: new insights to address complex dynamics after 30 years of developments. Ecol. Modell. 331 (July), 1–4. https://doi.org/10.1016/j.ecolmodel.2016.04.017.

Watari, S., Murase, H., Yonezaki, S., Okazaki, M., Kiyofuji, H., Tamura, T., Hakamada, T., Kanaji, Y., Kitakado, T., 2019. Ecosystem modeling in the western north pacific using ecopath, with a focus on small pelagic fishes. Mar. Ecol. Prog. Ser. 617–618 (May), 295–305. https://doi.org/10.3354/meps12508.

Wolf, P., 1992. Recovery of the Pacific Sardine and the California sardine fishery. Calif. Cooperative Oceanic Fish. Investig. Rep. 33, 78.

World Trade Organization, 2017. Fisheries Subsidies. Ministerial Decision of 13 December 2017. Buenos Aires. https://www.wto.org/english/thewto_e/minist_e/mc11_e/briefing_notes_e/bffish_e.htm.