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Abstract

The management of small pelagic fisheries is notoriously difficult due to environmental regime changes that generate multi-decadal cyclic fluctuations in stock abundance. Lagged management responses to environmental factors can amplify the effects of fishing and may even result in stock collapse. In this paper, we develop an age-structured bioeconomic model to explore the effectiveness of alternative management approaches for cyclically fluctuating small pelagic fish stocks. The fishery outcomes are evaluated against the overall profit of the fishery and the intertemporal distribution of fishing profits. The model is parameterised for the Japanese sardine fishery, once the largest fishery in Japan, which has experienced a prolonged period of stock collapse over the last 100 years. The results show that the duration of fishery collapse is mostly determined by the extent of cyclic fluctuations in the recruitment of immature sardines, but that the effects of the fluctuations on the fishery are heightened by the cumulative impact of fishing. We further show that restricting fishing reduces the fishery's overall profits, but smooths the intertemporal distribution of profits, resulting in greater intergenerational equity. This income smoothing effect is particularly pronounced when the stock exhibits high levels of cyclic fluctuations.

Keywords: bioeconomic models, environmental fluctuations, Japan, small pelagic fisheries, intergenerational equity

1. Introduction

The overall aim of this paper is to examine the fishery dynamics and effectiveness of alternative management approaches for small pelagic fisheries whose stocks are subject to cyclic fluctuations at multi-decadal timescales. The stocks of small pelagic fish, such as sardines and anchovies, are abundant relative to those of other marine fish species. Small pelagic species represent the world's largest species group, comprising around 30% of global marine capture production and contributing to 20% of global catch values of all marine fisheries (Pikitch *et al.* 2014; FAO 2016). Commercial exploitation of these species occurs globally at different scales, ranging from large-scale industrial fisheries to small-scale artisanal fisheries. The importance of these species to the economic and social development of coastal communities has long been recognised (Alder and Pauly 2008; Tacon and Metian 2009). Their benefits include direct human consumption and non-food uses, such as the production of fish meal and fish oil (Alder and Pauly 2008; Tacon and Metian 2009). Small pelagic fish also play a vital role in marine food webs by providing primary food sources for higher trophic level predators, such as marine mammals and seabirds (Cury *et al.* 2000; Pikitch *et al.* 2014).

Despite the broad benefits afforded by small pelagic fish stocks, the management of these stocks is notoriously difficult because species distribution, recruitment and growth are affected by abrupt environmental regime changes that occur at multi-decadal timescales (Beverton 1990; Kawasaki 1992; Fréon *et al.* 2005; Yatsu *et al.* 2005; Lindegren *et al.* 2013; Essington *et al.* 2015). The sensitivity of small pelagic fish to environmental conditions results in large cyclic fluctuations in abundance and, occasionally, stock collapse (Fréon *et al.* 2005). The magnitude and frequency of collapses of small pelagic fish stocks are amplified by fishing and lagged management responses to declining natural productivity (Beverton

1990; Essington *et al.* 2015). Moreover, multi-decadal fluctuations in stock abundance have serious economic and social implications, as evidenced by reports that fishing communities around the globe have been affected by the boom and bust cycle of small pelagic fisheries (Hamilton *et al.* 2006; Tacon and Metian 2009; Kim 2010).

In addition to environmental effects, small pelagic fisheries often experience overfishing, as these fish tend to form large, dense schools, allowing fisheries to remain economically viable even when abundance is low (Beverton 1990; MacCall 1990; Mackinson *et al.* 1997). The cyclic fluctuations in abundance also provide an incentive for fishers to invest in technology and capital when stock declines (Hamilton *et al.* 2006). Compared to other fisheries, however, small pelagic fisheries benefit less from the use of conventional static harvest strategies (e.g., an annual constant catch limit) or target reference points (e.g., maximum sustainable yield), since the relationship between the growth and abundance of the fish stock is unstable and varies over time (Carson *et al.* 2009; Hoshino *et al.* 2012). The literature has well established that stochastic fluctuations in resource dynamics undermine the effectiveness of fisheries management by exacerbating the consequences of overfishing and introducing additional decision-making complexity (Ludwig *et al.* 1993; Francis and Shotton 1997). An extensive body of bioeconomic research has also explored the implications of stochastic environmental variations of the optimal harvest strategy. Seminal studies in this area include the works of Clark and Kirkwood (1986), Reed (1979), Sethi *et al.* (2005), Weitzman (2001) and Costello and Polasky (2008).

However, few studies have examined the management implications of abrupt environmental regime changes at multi-decadal timescales. An exception is the work of Carson *et al.* (2009), who extended the conventional Gordon-Schaefer model to incorporate cyclic growth in stock dynamics. They found that a time-invariant harvest strategy generates resource rents that are

substantially lower than the economic optimum and that such a strategy may even result in a collapse of the fish stock. Another exception is Polasky *et al.* (2011), who examined the optimal harvest strategy when stock dynamics are subject to regime shifts. They showed that the possibility of a regime shift in stock dynamics makes the optimal harvest strategy more precautionary than when there is no regime shift.

In this paper, we particularly focus on the relationship between the net present value (*NPV*) of the fishery and the intertemporal distribution of fishing profits when fishery outcomes are evaluated under different management scenarios. We also explore how these measures of fishery performance are related to the possibility and extent of stock collapse in small pelagic fisheries. The *NPV* of the fishery has traditionally been used to evaluate the economic performance of alternative management plans (Grafton *et al.* 2010). By comparison, the intertemporal distribution of fishery profits has been less commonly incorporated in the formulation of management strategies; however, it is an important measure of fishery performance (Sumaila 2004). A significant disparity in the distribution of fishery benefits at different points in time may contribute to transient food insecurity or a boom and bust of fishing communities that rely heavily on small pelagic fish resources. However, such implications of resource management are not reflected in the *NPV* of fishing, which simply represents the sum of a fishery's intertemporal economic profits. We hypothesise that: (1) trade-offs between these key measures of fishery performance are inevitable in the management of small pelagic fisheries due to the presence of multi-decadal cyclic fluctuations and (2) the presence of such trade-offs is closely related to the possibility and extent of stock collapse.

To address these research objectives, we develop a bioeconomic model for a small pelagic fishery. The bioeconomic model developed in this paper incorporates the age-structured

population dynamics (Quinn and Deriso 1999; Haddon 2010), and the modified Ricker-type stock–recruitment relationship, which takes into account the characteristics of both density dependence and cyclic fluctuations (Anderson and Seijo 2011). We also characterise the fishery's baseline fishing behaviour using the non-parametric *k*-nearest neighbours regression (James *et al.* 2013). We evaluate the effectiveness of two management approaches—namely, proportional reduction of fishing mortality from the baseline level and the closure of nursery grounds—by comparing them with the simulation results of the baseline scenario.

The bioeconomic model developed in this paper is simulated with the estimated parameter values from the Japanese sardine Pacific stock (*Sardinops melanostictus*) fishery. The Japanese sardine Pacific stock fishery provides a useful case study for the research objectives. The fishery was once one of the largest fisheries in terms of volume of catches, accounting for around 30% of the marine capture production in Japan (Ichinokawa *et al.* 2017). It has long been known that the stock of Japanese sardines is sensitive to environmental conditions and exhibits drastic cyclic fluctuations in abundance at multi-decadal timescales (Kawasaki 1992). Considerable effort has been devoted to examining the influence of various environmental factors on the fluctuation of Japanese sardine stocks (Watanabe *et al.* 1995; Wada and Jacobson 1998; Noto and Yasuda 1999; Suda *et al.* 2005; Yatsu *et al.* 2005); however, there remains a lack of knowledge concerning how cyclic fluctuations in stock abundance relate to the fishery's economic and socioeconomic outcomes.

2. Bioeconomic model of a small pelagic fishery

2.1. Population dynamics

We apply the age-structured population dynamic model which has been used extensively in

the literature (Quinn and Deriso 1999; Haddon 2010; FRA 2014). In this model, the number of individuals for age-class $a = 1$ to one year before the maximum age-class A (i.e., $a = A$-1) is written as:

$$N_{a,t} = N_{a-1,t-1} \exp\left(-M - F_{a-1,t-1}\right) \text{ if } 1 \leq a \leq A - 1 \tag{1}$$

where $N_{a,t}$ is the number of fish population of age-class a in year t, M is the natural mortality, and $F_{a,t}$ is the fishing mortality at age-class a in year t. For the maximum age-class $a = A$, the number of individuals is given by:

$$N_{A,t} = N_{A-1,t-1} \exp\left(-M - F_{A-1,t-1}\right) + N_{A,t-1} \exp\left(-M - F_{A,t-1}\right) \text{ if } a = A \tag{2}$$

The catch in numbers for each age-class is given by the Baranov (1918) and Beverton and Holt (1957) catch equation:

$$C_{a,t} = \frac{F_{a,t}}{F_{a,t} + M} N_{a,t} \left[1 - \exp\left(-M - F_{a,t}\right)\right] \tag{3}$$

The empirical evidence suggests that the recruitment of small pelagic species is both density-dependent and subject to significant cyclic fluctuations due to environmental factors, such as changes in sea surface temperature and El Niño/Southern Oscillation (Kawasaki 1992; Sakuramoto 2013). Given these characteristics, we applied Anderson and Seijo (2011) recruitment equation, which incorporates the effects of both environmental fluctuations and density dependency into a modified Ricker equation:[1]

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[1] Conventional or modified versions of the Ricker equation are commonly used to model the recruitment of sardines and other pelagic species (Murphy 1967; Wada and Jacobson 1998; McClatchie *et al.* 2010; Rosa *et al.* 2017).

$$N_{0,t} = SSB_t \times \exp\left[\varphi_1 - \varphi_2 SSB_t - \varphi_3 \sin\left(2\pi \frac{t}{cycle}\right)\right] + \varepsilon_t^{N_0} \tag{4}$$

where SSB_t is the spawning stock biomass in year t, sin is a trigonometric function, φ_i (i = 1, 2, 3) are recruitment parameters, *cycle* is the length of the cycle that determines the nature of environmental fluctuations in recruitment and $\varepsilon_t^{N_0}$ is the error term. The spawning stock biomass depends on the number of mature individuals in the cohort, such that $SSB_t = \sum_{a=0}^{A} \mu_a N_{a,t} w_a$, where μ_a and w_a are the maturation rate and weight at age-class a, respectively.

The biomass and yield in year t can be calculated as:

$$B_t = \sum_{a=0}^{A} N_{a,t} w_a \tag{5}$$

$$Y_t = \sum_{a=0}^{A} C_{a,t} w_a \tag{6}$$

where B_t and Y_t denote the biomass and yield in weight in year t, respectively.

2.2. Fishery profit

Given the biomass and yield in (5) and (6), the economic profit of the fishery in year t is given by:

$$\pi_t = \pi(Y_t, B_t) = P(Y_t)Y_t - C(Y_t, B_t) \tag{7}$$

where $P(Y_t)$ is the inverse demand function and $C(Y_t, B_t)$ is the total cost function. Following

Clark (1990) and Grafton *et al.* (2000), we specify the inverse demand and cost functions as follows:

$$P(Y_t) = a_1 Y_t^{a_2} \exp(\varepsilon_t^p) \tag{8}$$

$$C(Y_t, B_t) = b_1 \frac{Y_t}{B_t} + b_2 + \varepsilon_t^c \tag{9}$$

where a_1, a_2, b_1 and b_2 are price and cost parameters, respectively, and ε_t^p and ε_t^c are error terms. Given the profit function in (7), the *NPV* of the fishery is given by:

$$NPV = \sum_{t=0}^{T} \pi_t (1+\delta)^{-t} \tag{10}$$

where δ is the time discount rate.

2.3. Fisher behaviour: baseline fishing mortality

To account for fishers' behaviours in changing environmental conditions, we applied the *k*-nearest neighbours (*k*NN) regression (James *et al.* 2013) to estimate time-varying baseline fishing mortality. The *k*NN regression is a non-parametric method that uses a training dataset of inputs to calculate the average of the *k* closest neighbours as an output value. More particularly, to calculate the baseline fishing mortality at year τ ($F_{a,\tau}^{BASE}$), we first constructed a training dataset by calculating the Euclidian distances between the simulated population size at year τ ($N_{a,\tau}$) and the historical series of population sizes at each age-class ($N_{a,(i)}$): that is, $\left\| N_{a,\tau} - N_{a,(i)} \right\|$, $i \in \{1, 2, ..., I\}$, where *I* is the size of the training dataset. We then sorted the

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training dataset in descending order: that is, $\left\| N_{a,\tau} - N_{a,(1)} \right\| \leq \left\| N_{a,\tau} - N_{a,(2)} \right\| \leq \cdots \leq \left\| N_{a,\tau} - N_{a,(I)} \right\|$.

Given this training dataset, we selected the shortest to the *k*-th shortest distances and matched them to the corresponding values of fishing mortality: that is, $\{ F_{a,(1)}, F_{a,(2)}, ..., F_{a,(k)} \}$, where k

$\leq I$. Finally, the baseline fishing mortality at age-class *a* in year τ was estimated by calculating:

$$F_{a,\tau}^{BASE} = \frac{1}{k} \sum_{k'=1}^{k} F_{a,k'}$$ (11)

We note that fishers' behaviour estimated by equation (11) is myopic, or not forward-looking, in that the fishing mortality in year *t* is determined based on the historical series of population sizes and fishing mortality in years $\tau < t$ only.

2.4. Management scenarios

In addition to the baseline scenario, we consider management scenarios in which fishing mortality is controlled by two alternative approaches, both of which are commonly used in the management of small pelagic fisheries. The first approach is a proportional reduction in fishing mortality from the baseline level. In this approach, we consider a case in which the rate of fishing mortality at each age-class is reduced in proportion to the baseline level: that is, $F_{a,t} = \phi F_{a,t}^{BASE}$, where $\phi < 1$. We assume in this scenario that fishing gear affects the mortality of all age classes equally (non-selective) because it is not possible to control the fishing mortality of particular age classes solely by limiting the total catch for the fishery, given the schooling behaviour of small pelagic species as well as the low selectivity of

fishing gear commonly used in small-pelagic fisheries (i.e., purse seines and set-nets).

The second management approach considered is a closure of nursery grounds to protect immature fish from harvesting.[2] For simplicity, we assume that when fishing is not permitted in the nursery grounds, the fishing mortality of the immature age-class becomes zero, while the fishing mortality of other age classes remains unaffected: that is, $F_{a,t} = 0$ for the immature stock, while $F_{a,t} = \phi' F_{a,t}^{BASE}$ for the mature stock and $\phi' \leq 1$.[3]

2.5. Measuring the duration of the stock collapse period and the intertemporal distribution of fishery profit

The outcomes of the baseline and the two alternative management approaches are evaluated using three performance indicators. First, the duration of the stock collapse period (*SCP*) is used as an indicator corresponding to the conservation outcome of fisheries management. We calculate the collapse period of the fishery as the total number of years during which the biomass is below a threshold of fish biomass, $\hat{B}$; that is:

$$SCP = \sum_{t=1}^{T} \Im\left(B_t < \hat{B}\right) \tag{12}$$

where T is the number of simulation years and $\Im$ is the indicator function, taking the value of

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[2] An alternative approach commonly used to protect immature fish is to limit the mesh size of the nets that can be used in the fishery. However, such gear restrictions are likely to be less effective in small pelagic fisheries, given the low selectivity of purse seine nets (FAO 1984).

[3] We should note here that, in practice, not all immature stock is likely to be protected by the closure of nursery grounds, as the distribution of immature stock is not fixed. For example, the Japanese sardines migrate along the east coast of Japan from January to March before settling in the nursery grounds.

one when the biomass is below the threshold, $\hat{B}$, and zero otherwise.

We also use two indicators that respectively correspond to the economic and socioeconomic outcomes of fisheries management: the *NPV* and the intergenerational equity or distribution of fishing profits over time.[4] To quantify the intertemporal distribution of fishing profits, we use the adjusted Gini coefficient (*GC*) proposed by Deltas (2003); that is:

$$GC = \frac{T}{(T-1)} \frac{\sum_{t=1}^{T}\sum_{t'=1}^{T}|\pi_t - \pi_{t'}|}{2T^2\bar{\pi}} \tag{13}$$

where $\bar{\pi} = T^{-1}\sum \pi_t$. The *GC* is bounded between zero and one, indicating perfect equality (i.e., the same profit is earned for all periods) and perfect inequality (i.e., a profit is realized in only one year).

3. Empirical application: the Japanese sardine Pacific stock fishery

3.1. Fishery background

The population of Japanese sardines (*Sardinops melanostictus*) has two major stocks: one on the Pacific side of Japan and the other in the Tsushima warm current region. The assessment of each stock is conducted separately. In this paper, we focus on the Pacific stock of Japanese sardines. The distribution of the Pacific stock is mainly associated with how the stock is transported by Kuroshio and Oyashio ocean currents at different life stages (Fig. 1). The

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[4] The literature has identified a range of socioeconomic benefits from fisheries and proposed various socioeconomic indicators, including indicators for community capacity and landing risk (Lundquist and Granek 2005; Rosa *et al.* 2017)

sardine migrates north during the spring and summer for feeding and south during the winter for spawning. Its main spawning grounds are in the Kuroshio current of southeast Japan. After hatching, the juveniles are transported to the nursery grounds in the Kuroshio and Oyashio transition zone in the northeast. The feeding grounds are in the Oyashio current, which stretches from the north of Japan to the southern areas of the Okhotsk Sea (Nishida 2005; FRA 2014). The main fishing gears used in the Japanese sardine Pacific stock fishery are large- and medium-sized purse seines and set-nets, which are used along the coast off Central to Northern Japan and along the inshore waters off southern Japan, respectively. Thus, the fishing grounds for the large- and medium-sized purse seines partly coincide with the nursery grounds (Yatsu *et al.* 2005; FRA 2014). The total purse seine catches of sardines account for 70 to 90% of the fishery's total yield.

[Figure 1 about here]

Figure 2 presents the biomass, yield and exploitation rate (i.e., the proportion of biomass removed by fishing per year) dynamics in the Japanese sardine Pacific stock fishery from 1976 to 2013. The figure also presents the sea surface temperature (SST) anomaly in the Kuroshio Extension for the same period. The biomass and yield increased constantly from 1976 until they peaked in the late 1980s. In the early 1990s, however, the fishery experienced a rapid decline in both biomass and yield, which declined by 89 and 80%, respectively, from 1990 to 1994. Previous studies suggest that these declines were associated with abrupt changes in climate and oceanographic conditions, resulting in multiple years of recruitment failure from 1988 to 1991 (Noto and Yasuda 1999; Nishida 2005; Yatsu and Kaeriyama 2005; FRA 2014). This is in part reflected in the Kuroshio Extension SST anomaly in these years (Fig. 2d)

[Figure 2 about here]

As one would expect, the fishery's exploitation rate is closely associated with stock abundance. Until the end of the 1980s, when there was an increasing trend in biomass, the exploitation rate remained stable at around 15% (Fig. 2c). After the stock collapsed, the exploitation rate became volatile (15–58%), and the mean exploitation rate increased to 35%. The higher rate of exploitation in the years of stock collapse reflects the increase in fishing pressure on the stock and the depletion of resources available for exploitation (FRA 2014).

Fisheries resources in Japan have traditionally been managed through a community-based management system with fishing rights and licenses adminstrated by the local Fisheries Cooperative Association or prefectual government (Yamamoto 1995; Makino and Matsuda 2005; Uchida and Makino 2008; Makino 2011). In 1997, in response to nationwide concern regarding the general health of coastal fishery resources and the ratification of the United Nations Convention on the Law of Sea (UNCLOS), a total allowable catch (TAC) system was introduced for seven species, including the Japanese sardine. Under the current management system, the Ministry of Agriculture, Forestry and Fisheries (MAFF) sets and announces the TAC level annually in accordance with stock assessments and estimates of allowable biological catch (ABC) reported by the Fisheries Research Agency (FRA). Along with the biological assessment, the socioeconomic conditions of the fishery and advice from the Fishery Policy Council, which is an advisory body to the government for national level fishery policy, are incorporated in the determination of the TAC (Makino 2011). After the TAC is set at the national level, the TAC is allocated to Fisheries Management Organisations (FMOs), which manage offshore fisheries, and prefectural governments, which administrate coastal fisheries (Makino and Matsuda 2005; Yagi and Managi 2011).

The effectiveness of TAC systems generally depends on whether: (1) the actual catch in the fishery does not exceed the TAC and (2) the TAC is binding or close to binding (Grafton *et al.* 2006; Kompas and Gooday 2007). When the TAC is non-binding, the fishery's yield or effort is, in effect, not constrained by the TAC (Emery *et al.* 2014; Rust *et al.* 2017). The TAC for the Japanese sardine has not been binding in any year since the management system was introduced to the fishery (FA 2017). This raises questions regarding the effectiveness of the current management system in reducing fishing pressure on the stock.[5]

3.2. Parameterisation and model validation

The parameter values used in the simulations are presented in Table 1. The natural mortality and maturity at age are taken from the stock assessment report (FRA 2014). The maximum age-class is set $A = 5+$ years old. The recruitment parameters, φ_i, and *cycle* are estimated using stock assessment data on the number of populations at each age-class, biomass and yield from 1976 to 2013 (Table 2). More particularly, we estimate the recruitment equation (4) for a given length of cycle between 0 and 100 years and choose the final model based on the Akaike Information Criterion (AIC). The threshold of biomass ($\hat{B}$) below which the stock is considered collapsed is set at five million tonnes, corresponding to the level of biomass in 1992 at the end of four consecutive years of recruitment failures (Nishida 2005; FRA 2014).

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[5] While a non-binding TAC has occurred in other TAC-managed fisheries globally (Grafton *et al.* 2007; Pascoe *et al.* 2007; Rust *et al.* 2017), it is an open question as to what caused the TAC to be non-binding for the Japanese sardine fishery. Given both biological and socioeconomic factors are considered to determine the TAC, possible reasons for the TAC being non-binding include that stock biomass is overestimated in the biological assessment of the fishery or the TAC is set too high in order to reduce the short-term socioeconomic impacts on fishing industry. For example, of seven TAC-managed species in Japan, there is no species that the TAC was binding in 2015 (FA 2017).

The inverse demand and cost functions (8) and (9) are estimated using data on the price of Japanese sardine per kilogram (1991–2013) and the total expenditures of large- and medium-sized purse-seine vessels (1994–2013). These price and cost data were retrieved from the annual survey reports on fisheries' business management published by the (MAFF 2016).[6]

[Table 1 about here]

[Table 2 about here]

Using the initial conditions of the fishery in 1976, we validated the parameterised model by simulating the biomass and yield from 1977 to 2013 with the time-variant baseline fishing mortality, as described in Section 2.3. The results show that the model explains 95 and 91% of the variations in the reported biomass and yield data, respectively, demonstrating a good in-sample fit of the model (Fig. 3).

[Figure 3 about here]

4. Results

4.1. Biomass and yield under the baseline scenario

Under the baseline scenario, both the biomass and the yield are predominantly driven by cyclic fluctuations in recruitment (Fig. 4a, b). The length of each cycle in biomass and yield corresponds to the estimated length of *cycle* (36 years) in recruitment. The biomass and yield

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[6] A relatively poor fit of the cost equation is likely due to the small sample size, althought this is the maximum number of observations we could obtain at the time of this study. The availability of cost data is a major obstacle for applied bioeconomic research, and this issue is widly recognised in the literature (Pascoe *et al.* 2014; Emery *et al.* 2017; Hoshino *et al.* 2018). We examined the sensitivity of our results to changes in the cost parameters (b_1 and b_2) and found that a ±20% change in the parameter values does not change our results qualitatively.

are not, however, fully independent from fishing. This is evident from our results, which

show that the biomass in the baseline scenario is consistently lower than the biomass when

there is no fishing (Fig. 4c). Moreover, the biomass periodically declines below the threshold

of five million tonnes, suggesting that the Pacific stock of Japanese sardines regularly

collapses when fishing pressure is maintained at the current level (see the shaded periods in

Fig. 4). Notably, stock collapse occurs even when there is no fishing; however, the presence

of fishing at the baseline level lengthens the stock collapse period. The mean duration of the

stock collapse period under the baseline scenario is 49 years, or 62% of the total simulated

period, whereas the collapse period when there is no fishing is 31 years, or 39% of the total

period.

[Figure 4 about here]

As one would expect, the yield in the fishery is lower during the collapse period than during

the non-collapse period (Fig. 4b). Despite the small amount of fishing occurring during the

collapse period, the difference in biomass between the baseline and no-fishing scenarios

during this period is particularly high (Fig 4c). For example, the average difference in

biomass between the two scenarios is 3.0 million tonnes during the collapse period, compared

to 2.1 million tonnes during the non-collapse period. The relative difference in biomass

between the collapse and non-collapse periods is driven not solely by the cyclic fluctuation in

recruitment, but also by the behavioural response of fishing fleets. This is reflected in

changes in the exploitation rate (Fig. 4d). The exploitation rate during the stock collapse

period is relatively stable, at around 30%, but decreases to around 10% at the end of the

collapse period. The decrease in the exploitation rate reflects a response to the gradual

recovery of the stock at the end of the collapse period. By contrast, the exploitation rate

steadily increases from 10 to 25% during the non-collapse period.

Changes in the exploitation rate, along with cyclic fluctuations in recruitment, also affect the relative abundances of mature and immature stock over time. For example, the biomass of immature stock is relatively high at the beginning of the non-collapse period (Fig. 4a). However, as the exploitation rate increases, the relative abundance of immature stock declines. The decline in the immature stock, combined with a further increase in the exploitation rate, contributes to a depletion of mature stock towards the end of the non-collapse period. Once the stock collapses, an average of 42% of the yield is immature stock. This proportion is significantly higher than that during the non-collapse period (25%). The high proportion of immature stock in the yield contributes to the prolonged duration of the collapse. When multiple years of good recruitment occur in cyclic fluctuations, the immature stock rebuilds first, and the mature stock biomass increases later.

4.2. Effects of controlling fishing mortality and area closure

Figure 5 shows the relative effects of a proportional reduction in fishing mortality from the baseline level and the closure of nursery grounds to protect immature sardines. The baseline case is one in which the fishery maintains fishing mortality at the baseline level for 80 years of simulation.

[Figure 5 about here]

The duration of the stock collapse period consistently decreases with a reduction in fishing mortality (Fig. 5a). However, shorter stock collapse durations are achieved only at the expense of decrease in the *NPV* (Fig. 5b). For example, a 30% reduction in fishing mortality

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relative to the baseline level results in a 6% decrease in the stock collapse period, but a 14% decrease in the *NPV*. Preventing the harvest of immature sardines through the closure of nursery grounds creates the same trade-off between a shorter period of stock collapse and a lower *NPV*. For example, when fishing mortality is reduced to 70% of the baseline level in conjunction with area closure, the stock collapse period and the *NPV* decrease by 15 and 22%, respectively.

Regardless of whether area closure is applied, a reduction in fishing mortality appears to improve equity in the intertemporal distribution of fishery profits. For example, reducing fishing mortality to 70% of the baseline level results in a 10% decrease in the Gini coefficient, suggesting a more equal distribution of the fishery's economic profits over time (Fig. 5c). For a given level of fishing mortality, however, the closure of nursery grounds to fishing lowers equity in the intertemporal distribution of fishery profits.

4.3. Profit and equity trade-offs

Our results suggest that there are trade-offs among performance indicators when fishing mortality is reduced from the current baseline level, regardless of area closure. To evaluate the extent of these trade-offs, we calculate the trade-off ratio of a percentage change in the *NPV* to one percentage change in the stock collapse period (i.e., $\eta_{SCP} = \%\Delta NPV/\%\Delta SCP$), as well as to one percentage change in the Gini coefficient (i.e., $\eta_{GC} = \%\Delta NPV/\%\Delta GC$). Calculating the trade-off ratios η_{SCP} and η_{GC} is useful. For example, η_{SCP} represents the responsiveness of the fishery's economic performance, measured by the *NPV*, to a change in the conservation outcome, measured by the duration of the stock collapse period. A positive trade-off ratio ($\eta_{SCP} > 0$) implies that a decrease in the duration of the stock collapse period

due to a reduction in current fishing mortality is associated with a decrease in the *NPV*. Furthermore, a trade-off ratio above one ($\eta_{SCP} > 1$) means that the proportional change in the *NPV* is greater than the change in the stock collapse period. Similarly, the ratio η_{GC} measures the responsiveness of the fishery's economic performance to a change in its socioeconomic outcome.

[Figure 6 about here]

When there is no area closure, the trade-off ratio η_{SCP} is always above one and remains stable at around two, regardless of the reduction in fishing mortality from the baseline level (Fig. 6a). This means that the proportional reduction in the duration of the stock collapse period due to reduced fishing mortality is less than the proportional reduction in the *NPV*. For example, a 30% reduction in fishing mortality from the baseline level yields 6 and 14% decreases in the duration of the stock collapse period and the *NPV*, respectively, so that $\eta_{SCP} =$ 2.3. This trade-off ratio between the *NPV* and the stock collapse period decreases when fishing mortality is controlled in conjunction with the closure of nursery grounds. For example, when fishing mortality is reduced to 70% of the baseline level and combined with area closure, the trade-off ratio is smaller ($\eta_{SCP} = 1.5$) than when fishing mortality is reduced to the same level with no area closure ($\eta_{SCP} = 2.3$).

The trade-off ratio between the *NPV* and the Gini coefficient also remains above one for all ranges of fishing mortality ($\eta_{GC} > 1$, Fig. 6b), suggesting that the decrease in the *NPV* caused by the reduced fishing mortality is proportionally greater than the decrease in the Gini coefficient (i.e., a more equal distribution of fishery profits over time). This result holds regardless of area closure; however, the trade-off ratio increases when fishing mortality is reduced in conjunction with area closure. For example, reducing fishing mortality to 70% of

the baseline results in a trade-off ratio of $\eta_{GC} = 1.3$. By comparison, the same reduction in fishing mortality in conjunction with area closure results in a trade-off ratio of $\eta_{GC} = 2.7$. The trade-off ratio is particularly high when the current level of fishing mortality is only moderately reduced in conjunction with area closure.

4.4. The role of cyclic fluctuations

To further explore the role of cyclic environmental fluctuations in the management of Japanese sardines, we examine the sensitivity of our simulation results to the strength of cyclic fluctuations. We accomplish this by proportionally varying the value of the parameter φ_3, which represents the strength of cyclic fluctuations in recruitment. Specifically, we re-simulate the model for 80 years with three alternative values of φ_3 in the recruitment equation (4): that is, $\tilde{\varphi}_3 = \omega \times \varphi_3$, where $\omega \in \{0.5, 1, 1.5\}$. The simulation results are presented in Figure 7.

[Figure 7 about here]

As shown in the baseline case (Fig. 5), the duration of the stock collapse period decreases with a reduction in fishing mortality for all values of φ_3 considered in this paper (Fig. 7a). However, the relative effects of reduced fishing mortality on the stock collapse period vary depending on the size of the cyclic fluctuations. In particular, the weaker the cyclic fluctuations, the greater the decrease in the duration of the stock collapse period for a given rate of fishing mortality. For example, when the parameter φ_3 is 50% of the baseline case (i.e., $\tilde{\varphi}_3 = 0.5\varphi_3$), a 30% reduction in fishing mortality results in a 9% decrease in the stock collapse period. By contrast, the same reduction in fishing mortality for the baseline case

(i.e., $\tilde{\varphi}_3 = \varphi_3$) yields only a 6% decrease in the stock collapse period. The qualitative nature of this result holds regardless of area closure (Fig. 7d).

Our results show that shorter durations of stock collapse periods are achieved only at the expense of a decreased *NPV* for all values of the parameter φ_3 (Fig. 7b, e). However, the relative change in the *NPV* differs depending on the size of the cyclic fluctuations. That is, a decrease in the *NPV* due to the reduction of current fishing mortality is smaller when cyclic fluctuations are weak. Similarly, when cyclic fluctuations are weak, the reduction in the Gini coefficient is less pronounced (Fig. 7c).

4.5. Risk-averse fisheries management

Our results show that a reduction in fishing mortality improves equity in the intertemporal distribution of fishery profits by mitigating the effects of environmental fluctuations on the fishery. This result implies that a preferred approach to the management of small pelagic fisheries may depend on the risk preference of fishery managers, in particular whether, and to what extent, the variability in fishery profits ought to be reduced at the expense of decreased *NPV* of the fishery. In other words, considering profit alone to evaluate the economic outcome of the fishery implicitly assumes that fishery managers are risk neutral. To examine how the extent of risk aversion affects the economic performance of different management approaches, we consider an alternative indicator; that is the sum of the discounted utility of fishing profits (i.e., intertemporal utility function) given as:

$$\sum_{t=0}^{T} \beta^t U\left(\pi_t\right) = \sum_{t=0}^{T} \beta^t \left(\frac{\pi_t^{1-\rho} - 1}{1-\rho} \right) \tag{14}$$

where $\beta = (1+\delta)^{-1}$ is the discount factor and ρ is the coefficient of relative risk aversion (Mas-Colell *et al.* 1995). The greater the value of ρ, the higher the risk aversion of fisheries management. We consider three alternative values of $\rho = 0$, 0.5 and 1.0, in which the simulation with $\rho = 0$ is equivalent to the baseline simulation reported in Figure 5.

[Figure 8 about here]

Figure 8 shows that utility decreases with a reduction in fishing mortality from the baseline level, regardless of area closure. This result is consistent with the result in Figure 5. However, the relative effects of reduced fishing mortality are weakened by increasing the coefficient of relative risk aversion. For example, a 30% reduction in fishing mortality results in a 14% decrease in the utility when a risk neutral fisheries management is assumed (i.e., $\rho = 0$). By contrast, the same reduction in fishing mortality results in only a 6 and 1% decrease in the utility when the coefficient of relative risk aversion is $\rho = 0.5$ and 1.0, respectively. These results suggest that, for the management of small-pelagic fisheries, risk-averse fishery managers prefer a greater restriction on fishing activity than risk-neutral managers do because the variability in fishery profits is less tolerated by risk-averse managers.

5. Discussion

The management of small pelagic fish stocks is notoriously difficult due to abrupt environmental changes that cause multi-decadal cyclic fluctuations in stock abundance. In this paper, we explore the fishery dynamics of the Pacific stock of the Japanese sardine and the effectiveness of different management approaches for this fishery. We focus particularly on the relationship between the fishery's overall profit and the intertemporal distribution of

fishery profits and how these performance measures are related to the duration of stock collapse. Understanding the associations between fishery's economic and socioeconomic outcomes and cyclic environmental fluctuations is highly relevant to many coastal countries, as small pelagic species represent the world's largest species group in terms of both catch and value.

Our simulation results show that the biomass and yield of the fishery are primarily driven by cyclic fluctuations in the recruitment of immature sardines. We find that the fishery would experience stock collapse around 40% of the time, even in the absence of fishing. This result is compatible with the fishery's actual trajectory over the last 100 years. However, this does not mean that the level of stock biomass—and, hence, the duration of the stock collapse period—is independent of fishing. Rather, our results suggest that if fishing mortality remains steady at the current rate, the average biomass will be 30% lower and the duration of the stock collapse period will be 60% longer, on average, than when fishing is ceased entirely.

The effects of environmental fluctuations on the fishery are further amplified by the behavioural responses of fishing fleets to changing environmental conditions. In particular, fishing has a cumulative impact through changes in the population's age structure. Our simulation shows that the proportion of immature fish in the total fishery catch is low when the stock is relatively abundant; however, once the stock collapses, the proportion of immature fish nearly doubles, prolonging the duration of the stock collapse period. These results are consistent with the fishery's stock assessment data and observations made for other small pelagic fisheries around the globe (Beverton 1990; Essington *et al.* 2015).

As one would expect, the duration of the stock collapse period in the Pacific stock of the Japanese sardine fishery can be shortened by either prohibiting fishing at the nursery grounds

or reducing fishing mortality by tightening the TAC. However, the effectiveness of these policy instruments is weakened when the stock exhibits high levels of cyclic fluctuations. In our baseline simulation case, fishing mortality needed to be halved to achieve a 10% reduction in the duration of the stock collapse period; however, when the extent of cyclic fluctuations was 50% lower than the baseline case, the same conservation outcome could be achieved by reducing fishing mortality by only 35%. Shortening the fishery's collapse period is not costless; rather, it is achieved only at the expense of the overall profit of the fishery. This result holds for all scenarios considered in this paper, implying a clear trade-off between improving conservation outcomes and sacrificing the economic returns from the fishery. Previous research has reported such trade-offs between multiple objectives in fisheries management (Mardle and Pascoe 1999; Hilborn 2007); however, this study is the first to demonstrate how the nature of the trade-offs is impacted by cyclic stock fluctuations.

Managing the duration of the stock collapse period is relevant to fisheries management not only for conservation purposes, but also for the intertemporal distribution of fishery benefits. Our results show that reducing fishing mortality from the current level in the Japanese sardine Pacific stock fishery will smooth fluctuations in fishery benefits, yielding greater intergenerational equity. Such smoothing effects are particularly pronounced when the stock exhibits high levels of cyclic fluctuations. This result reflects the fact that the unequal distribution of fishery profits is largely driven by cyclic fluctuations, coupled with the behavioural responses of fishing fleets to changing environmental conditions.

Although bioeconomic models conventionally set the maximization of overall fishery profits as the objective of fishery management, the optimal management policy derived from such a framework for a small pelagic fishery may yield a severely skewed distribution of fishing

profits or create a boom and bust of fishing communities. While restricting fishing activities may reduce the fishery's overall economic returns, this may be acceptable or even desirable when the boom and bust cycle is associated with substantial socioeconomic costs. For example, the boom and bust cycle of small pelagic fish resources has created several issues in fisheries management, such as overinvestment by fishers, the creation of fisher assistance programs that exacerbate overfishing in the long run and the spillover of fishing capacity to other fisheries (Hamilton *et al.* 2006; Tacon and Metian 2009; Kim 2010; Sumaila *et al.* 2010).[7] A prolonged bust period may also trigger transient food insecurity and substantial livelihood loss for community members who rely heavily on small pelagic fish resources. Moreover, the aggregated fishery profits for a given generation may not ensure the economic viability of the fishing community in the long run. In such situations, it is necessary to either restrict fishing during the boom or facilitate an assistance program to transfer the higher profits to compensate fishers during the prolonged stock collapse period. Overall, our results suggest that incorporating a measure of intergenerational equity in management decisions is imperative for small pelagic fisheries whose stock fluctuations create a boom and bust cycle at multi-decadal timescales.

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[7] Technological progress is another factor that may contribute to the boom and bust cycle of small pelagic fisheries and influence the intertemporal distribution of fishery profits. We unfortunately do not have data to measure the extent of technological progress in the Japanese sardine Pacific stock fishery; however, previous studies have quantified technological change and examined its implications to fisheries. For example, Hannesson *et al.* (2010) show that technological progress in the Lofoten cod fishery in Norway had been neutralised by a decline in the fish stock, so that the Lofoten fishers failed to enjoy the full benefits of technological progress. Squires and Vestergaard (2013) also show that the economically optimal stock is lower than what the conventional bioeconomic model (Clark *et al.* 1979; Grafton *et al.* 2007) suggests when technical progress is accounted for. We also note that major technological development in modern purse seining occurred in the 1950s (Whitmarsh 1990; Valdemarsen 2001; Torres-Irineo *et al.* 2014), which is outside the study period (1976-2013).

Acknowledgement

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Table 1. Parameter values

Parameter	Description	Value
M	Natural mortality rate[†]	0.4
$\mu_{a=1}$	Maturity at age-class 1[†]	0.1 if $B_t \geq 1,467$ 0.2 if $B_t \geq 758$ 0.5 if $B_t < 758$
$\mu_{a \geq 2}$	Maturity at age at age-class ≥ 2[†]	1.0
A	Maximum age-class[†]	5 or older
φ_1	Recruitment parameter[‡]	2.53
φ_2	Recruitment parameter[‡]	0.00023
φ_3	Recruitment parameter[‡]	-2.77
$cycle$	Recruitment cycle (years)[‡]	36
$\hat{B}$	Threshold of biomass below which the stock is considered collapsed (million tonnes)[†]	5
a_1	Price parameter[‡]	1.09E+09
a_2	Price parameter[‡]	-0.54
b_1	Cost parameter[‡]	-5.64E+08
b_2	Cost parameter[‡]	2.05E+09
δ	Time discount rate[†]	0.04

[†] The parameter values are taken from the stock assessment report of the Japanese sardine published by National Research Institute of Fisheries Science (FRA) in Japan
[‡] The parameter values are calculated or set by authors

Table 2 Estimation results of the recruitment, inverse demand and cost equation

	Equation (4) Recruitment		Equation (8) Inverse demand		Equation (9) Cost
φ_1	2.53 (0.000)	a_1	1.09E+09 (0.000)	b_1	-5.64E+08 (0.084)
φ_2	2.30E-04 (0.000)	a_2	-0.54 (0.000)	b_2	2.05E+09 (0.000)
φ_3	-2.77 (0.000)				
cycle	36 years				
Number of observations	38		21		17
Error variance	3.98E+04		0.27		2.31e+08
R^2	0.80		0.79		0.20

This table reports the estimates of the parameters in the recruitment, inverse demand and cost equations. The corresponding *p*-values are reported in parentheses. The recruitment and cost equations were estimated by OLS. The inverse demand equation was estimated using the censored regression model by truncating the prices at the observed minimum and maximum prices of JPY 22 and JPY 216 per kilogram

Figure 1. Distribution map of the Japanese sardine Pacific stock
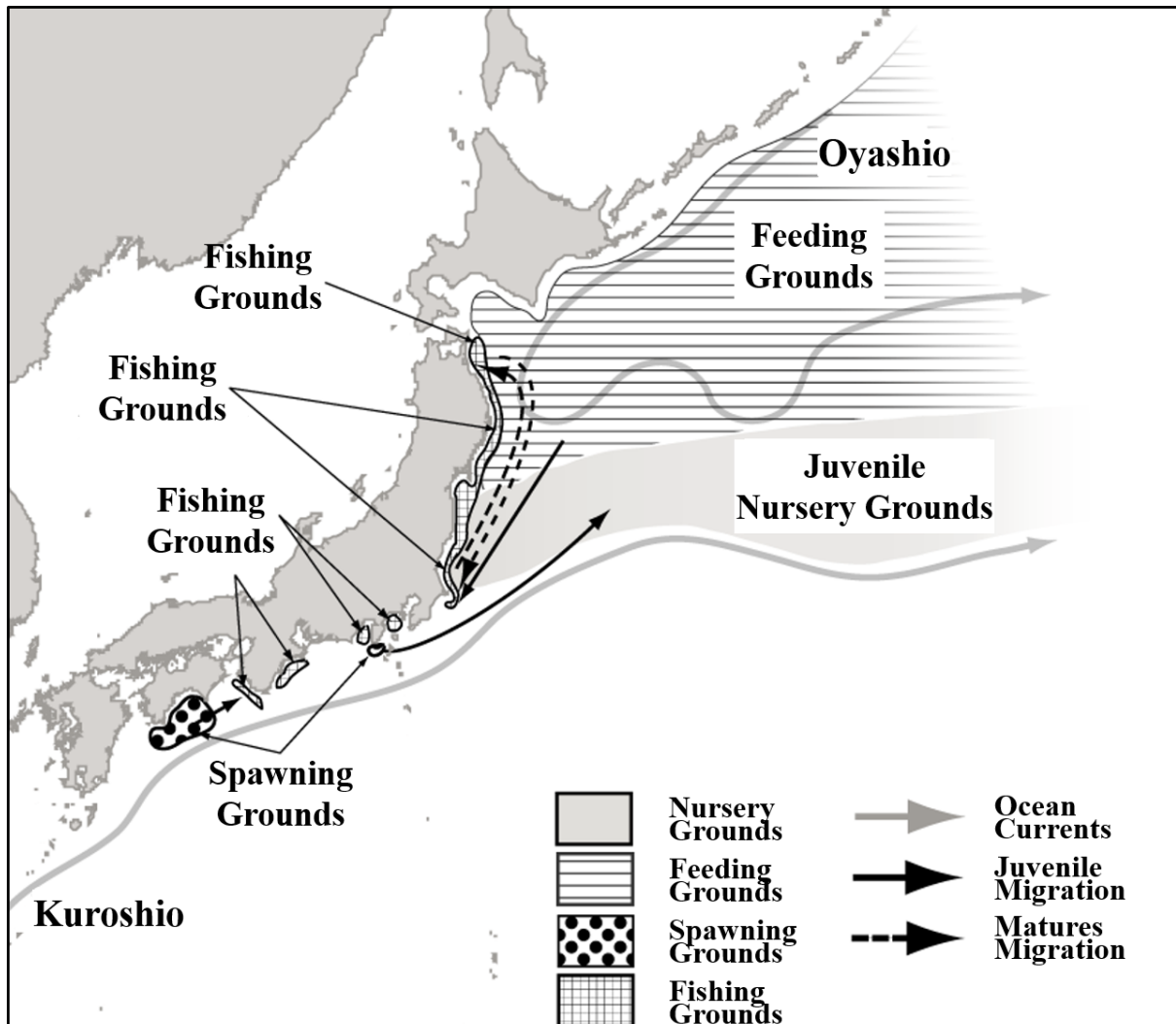
The Map is taken from FRA (2014) and modified by authors

Figure 2. Biomass (a), yield (b) and exploitation rate (c) in the Japanese sardine Pacific stock fishery, and the sea surface temperature (SST) anomaly in the Kuroshio Extension from 1976 to 2013. The immature stock consists of the age class 0 and 50% of the age class 1. The rest of the age-class form the mature stock.
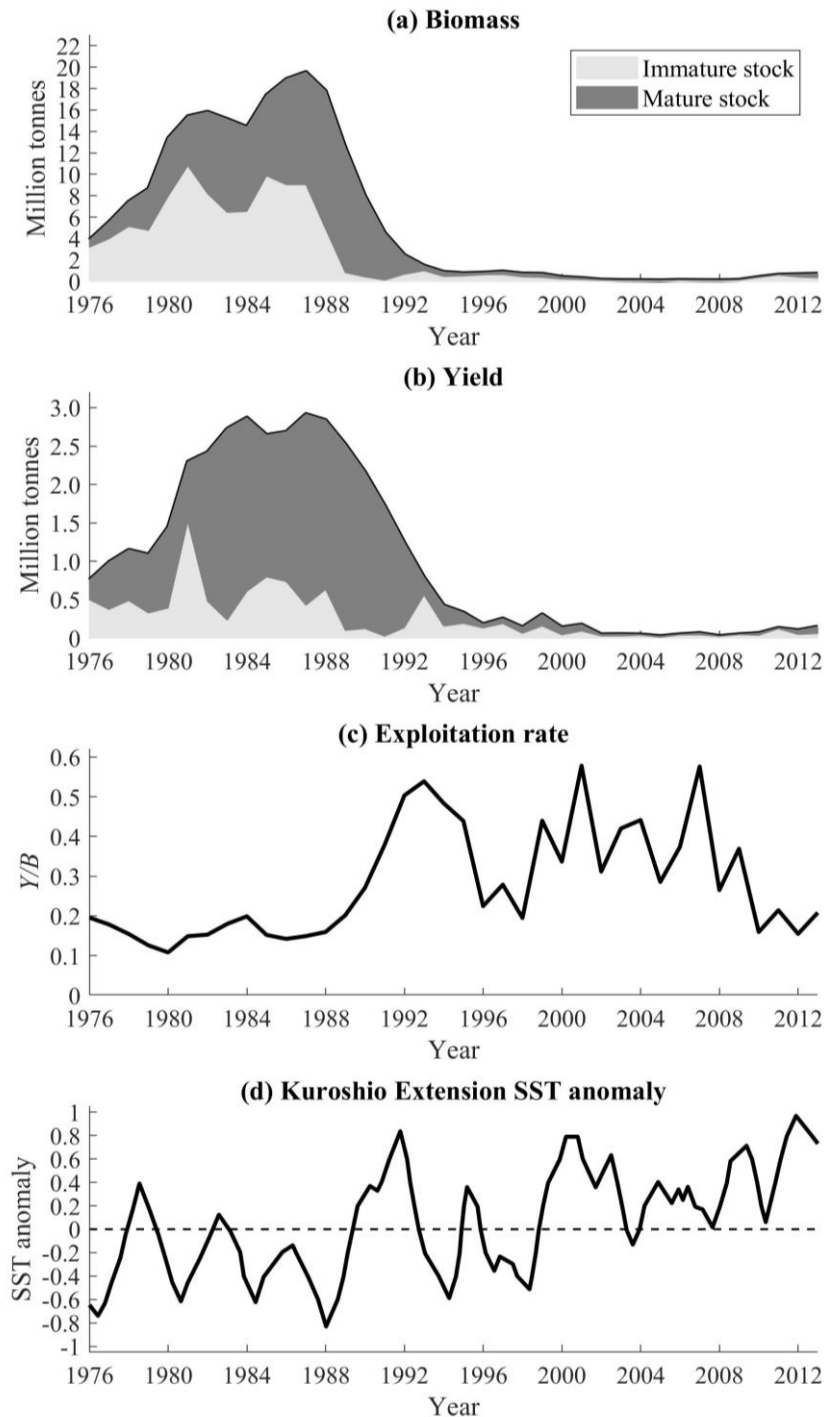
Figure 3. Model validation. Scatter plot of reported values in the stock assessment report (FRA 2014) against the values of biomass (a) and yield (b) predicted by the model. The grey line represents a regression line by least-squares.
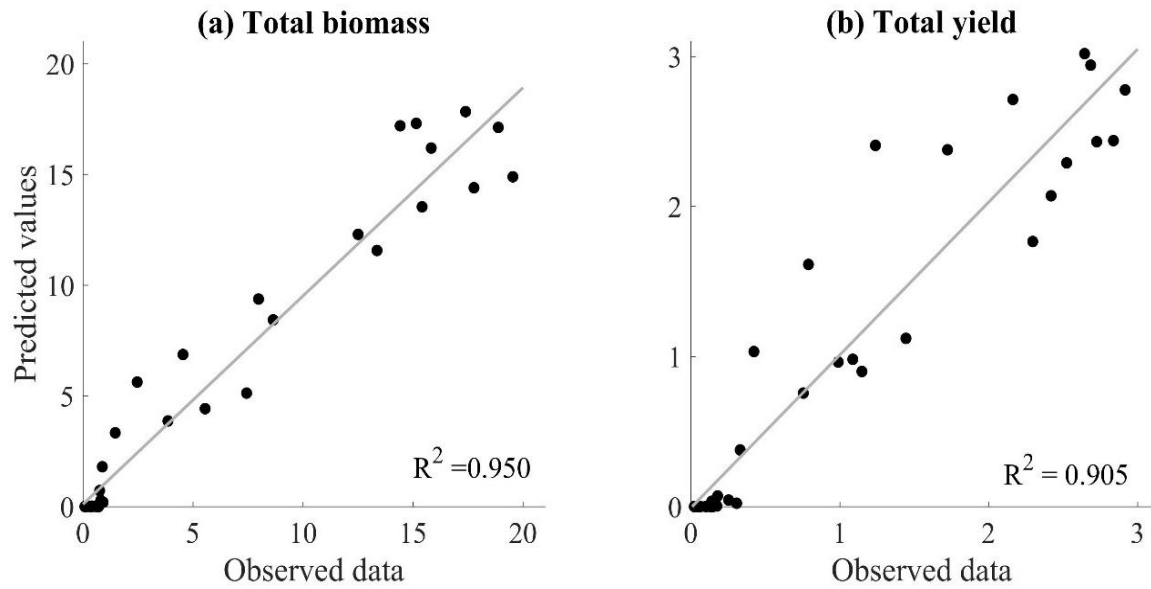
Figure 4. Baseline scenario: biomass (a); yield (b); difference in biomass between baseline and no-fishing scenarios (c); and exploitation rate (d). The blue areas represent the stock collapse period in which the biomass is less than five million tonnes. The results are based on the mean of 1,000 Monte Carlo samples.
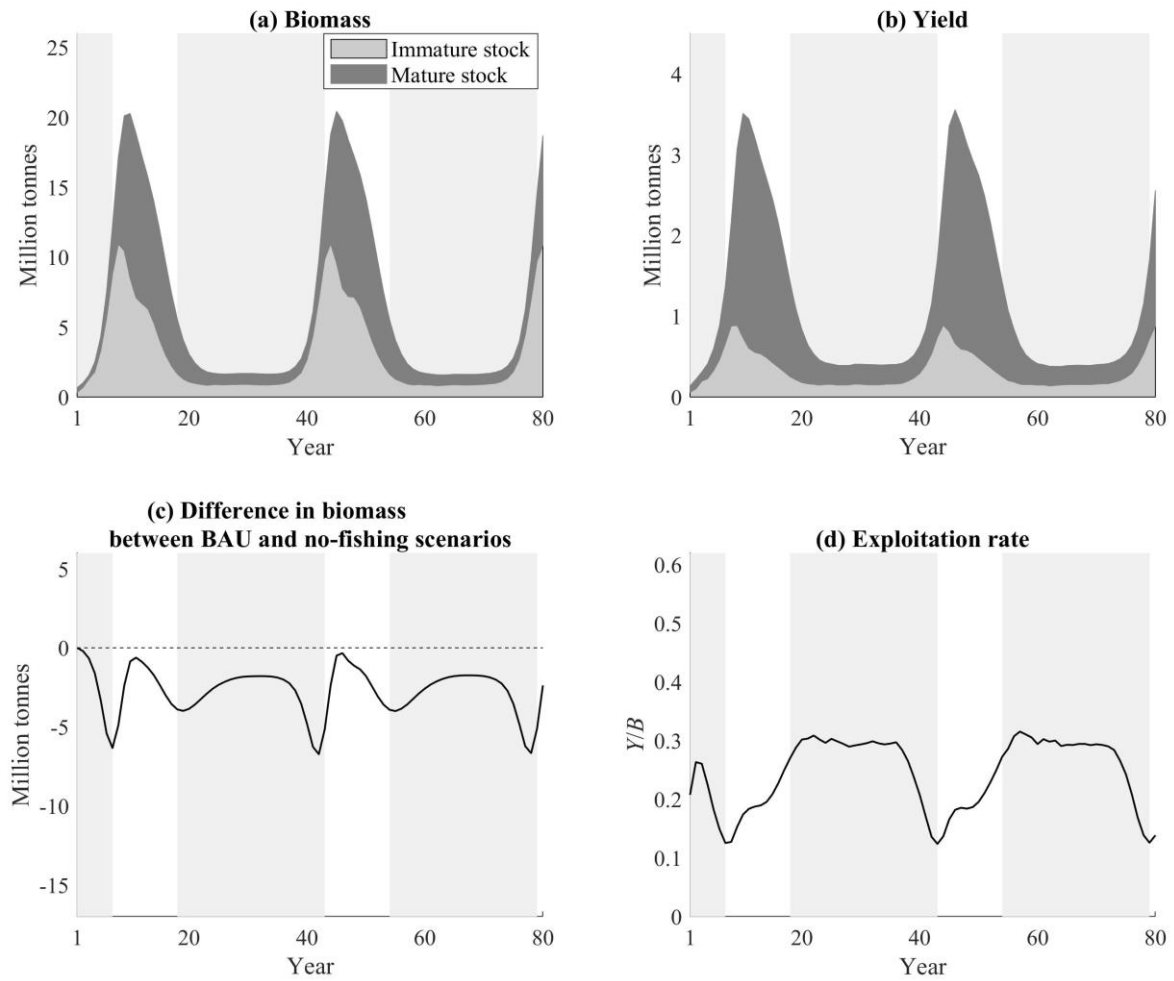
Figure 5. Relative effect of reducing fishing mortality without area closure (solid line) and with area closure (dashed line). The baseline case is $F = F^{BASE}$ with no area closure. Panel (a): stock collapse period. Panel (b) NPV of the fishery. Panel (c) Gini coefficient.
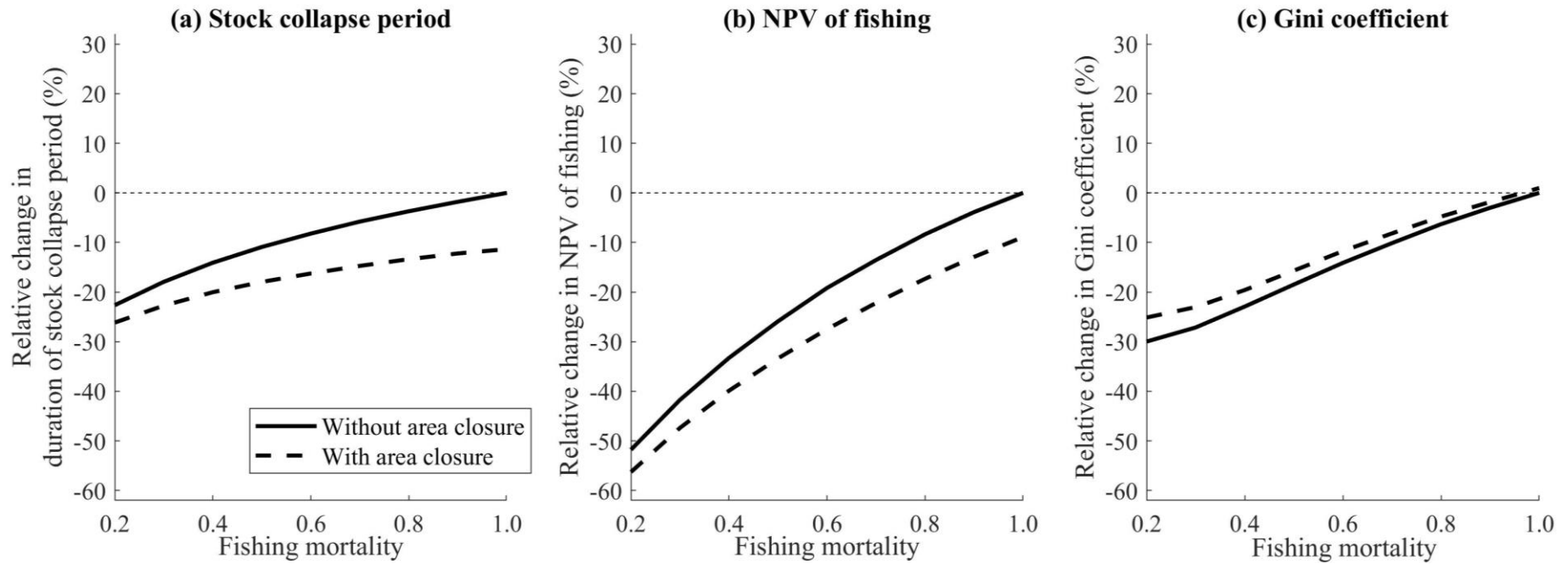
Figure 6. Trade-offs between the *NPV* of the fishery and stock collapse period (a) and between *NPV* and Gini coefficient (b). Trade-off effects of changes in fishing mortality without area closure (solid line) and with area closure (dashed line). The baseline case is $F = F^{BASE}$ with no area closure. The trade-off ratio is not calculated when the stock collapse period is longer than the baseline case or when the Gini coefficient is above the baseline case.
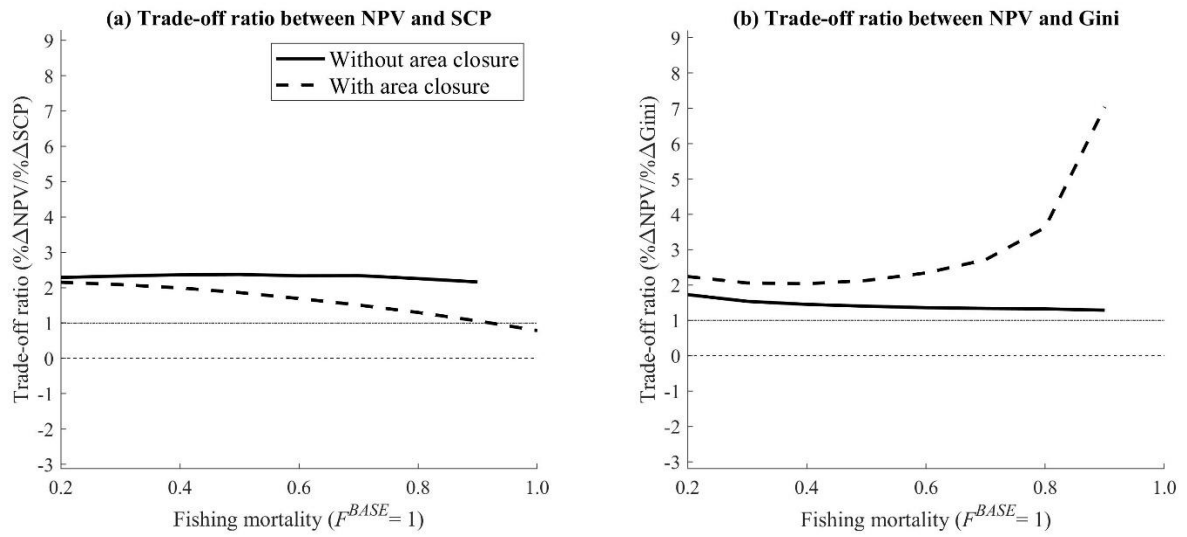
Figure 7. Effects of changes in the size of cyclic fluctuations in recruitment on the duration of stock collapse period (a, d), *NPV* of the fishery (b, e) and Gini coefficient (c, f). Five alternative values of φ_3 in the recruitment equation (4) are applied: $\tilde{\varphi}_3 = \omega \times \varphi_3$, $\omega \in \{0.5, 1, 1.5\}$. Fishing mortality is reduced relative to the baseline level, either with (a-c) or without (d-f) area closure. The baseline case is $F = F^{BASE}$ with no area closure.
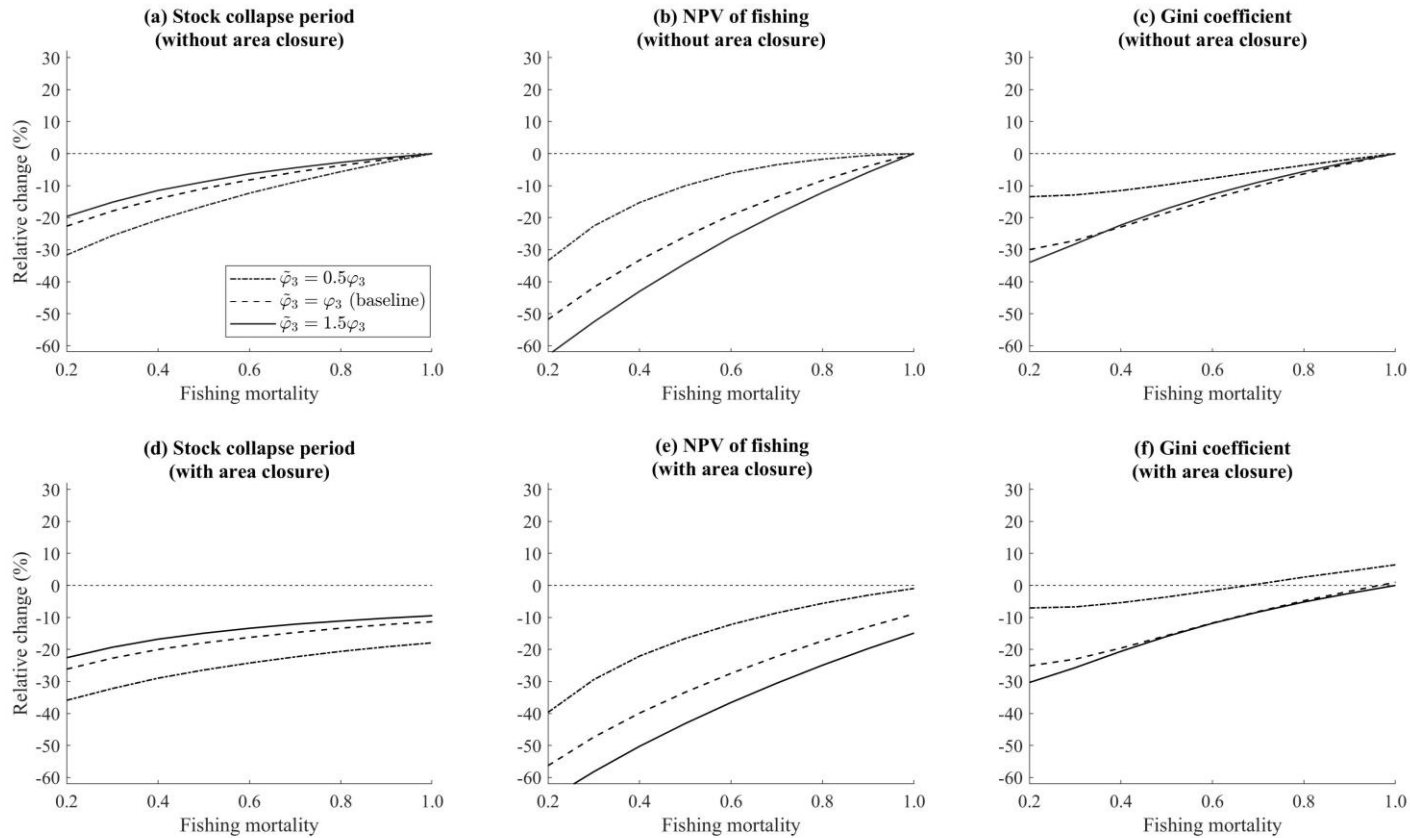
Figure 8. Relative changes in intertemporal utility for different coefficients of risk aversion (ρ = 0, 0.5 or 1.0) Fishing mortality is reduced relative to the baseline level, either with (a) or without (b) area closure. The baseline case is $F = F^{BASE}$ with no area closure.