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Quo Vadimus

Climate change and non-stationary population processes in fisheries management

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The potential influence of climate change on the future distribution and abundance of fish (and therefore commercial fisheries and food security) is increasingly recognized in the fishery management community. A changing climate will likely have differing effects on different species; some will flourish, some will flounder. Management targets for fishing mortality and spawning biomass are often calculated by assuming stationary population processes, but under climate change, this assumption may be violated. Non-stationary population processes can introduce bias into estimates of biomass from stock assessments and calculations of target fishing mortalities and biomasses. However, few accepted frameworks exist for incorporating the changing influence of the environment on exploited populations into management strategies. Identifying changes in population processes due to environmental influences is important in order to enable climate-enhanced management strategy evaluations to elucidate the potential benefits and costs of changing management targets. Cost/benefit analyses will also be useful when coincidentally caught species respond differently to environmental change.

Marine resources: demand and management

The demand for marine resources is high; at least 15% of worldwide dietary protein is derived from fish on average (Bènè *et al.*, 2015). Demand is likely to increase in the future, with global population projected to increase to 9 billion by 2050 (UN-DESA, 2009), but seafood supply from capture fisheries stabilized at 85–95 million metric tonnes (mmt) in the 2000s (FAO, 2009). If protein derived from the oceans is to remain at 20% of the diet of the added population, 75 mmt more protein will be required (from either capture fisheries or aquaculture) by 2050 (Rice and Garcia, 2011). Over the same period, oceans are projected to rise, warm, and become more acidic (Figure 1, Doney *et al.*, 2012). Exploited marine populations can be strongly influenced by environmental changes (Hollowed *et al.*, 2013a). For example, the recruitment of several commercially harvested species in the Bering Sea has been reported to be impacted by decadal climate variability [e.g. pollock (Mueter *et al.*, 2011), snow crab (Szuwalski and Punt, 2013), flatfish (Wilderbuer *et al.*, 2013), and salmon (Hare and Mantua, 2000;

Mundy and Evenson, 2011)]. Identifying harvest strategies that can sustain seafood production to satisfy the dietary needs of an expanding population under the myriad influences of climate change on fish production is critical.

Fisheries managers are faced with balancing a complex suite of interacting objectives representing competing needs of fishery-dependent communities and ecosystems in which an exploited population resides (Hollowed *et al.*, 2011). Decisions by fisheries managers reflect their (or their organizations') propensity for risk, expectations about the future, and the relative trade-offs between different ecosystem services (Ruckelshaus *et al.*, 2013). Defining risk in relation to a set of expectations about the future requires defining a frame of reference from which to evaluate performance (Prager *et al.*, 2003; Punt *et al.*, 2014a, b). Management performance is typically measured by how biomass and/or fishing mortality rates compare with a target or limit reference points, but reference points are implemented differently around the globe. For example, in the United States, both biomass and fishing mortality limits and
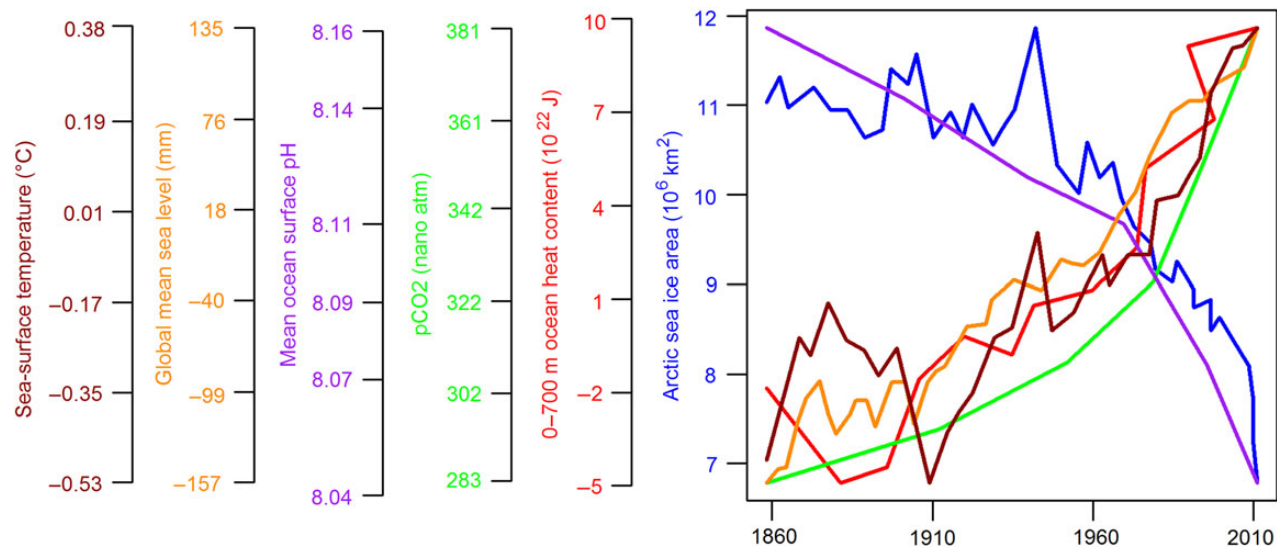
Figure 1. Time-series of climate indices relevant to ocean dynamics exhibiting non-stationary behaviour (data from Doney *et al.,* 2012).
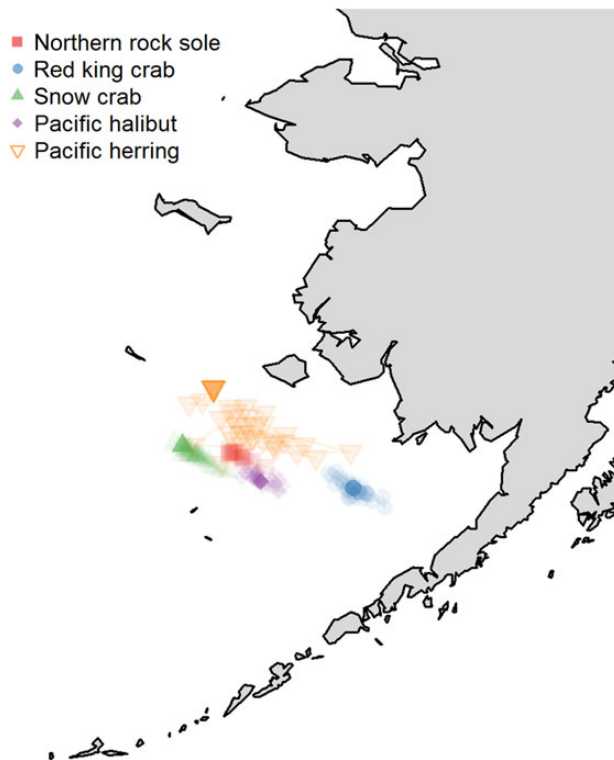

Figure 2. Movement in the centroids of biomass from the National Marine Fisheries Service (NMFS) summer survey for selected stocks in the Bering Sea. Each stock has a significant trend (*p* < 0.05) over time towards the north and west. Twenty per cent of populations in the Bering Sea showed a significant movement in at least one of the two directions. The darkest point indicates the most recent observation for each species. This figure is available in black and white in print and in colour at *ICES Journal of Marine Science* online.

targets are implemented. A US stock is considered "overfished" when biomass is some proportion (often 50%) of the target and "overfishing" is considered to occur when estimated fishing mortality is above the limit. In Europe, however, targets exist for fishing

mortality, but there are only "limits" for biomass, beneath which fishing is limited. Regardless of the locale, methods used to calculate reference points often assume stationary population dynamics and (often) a relationship between spawning biomass and recruitment. However, these assumptions have been shown to be uncertain for some stocks (e.g. Szuwalski *et al.*, 2015) and will likely need to be adjusted as ocean conditions change. Proxies for management targets (Clark, 1991) and the idea of "pretty good yield" (Hilborn, 2010) have been introduced to cope with the failure to precisely know the target biomass and the relationship between spawning biomass and future recruitment to the fishery. Fishing mortality rates that achieve a spawning biomass per recruit ratio of 40% of that at no fishing has been shown to produce yield close to maximum sustainable yield for a range of steepnesses (Clark, 1991). However, new frameworks to adjust these proxies to account for non-stationarity (i.e. changes in the mean or variance of a process over long periods of time) in population processes such as shifts in reproductive success, mortality, growth, or maturation rate under climate change will be needed.

Examples of changes in population processes induced by environmental changes

Non-stationarity in population processes can result from linkages to non-stationary environmental variables (Fulton, 2011). For example, changes in availability can occur when distributions of species shift in response to changes in environmental conditions. These shifts have occurred pole-ward in response to warming (e.g. Shackell *et al., 2012*; Poloczanska *et al.*, 2013), but not all species respond uniformly (Hollowed *et al.*, 2013b; Kotwicki and Lauth, 2013). Pinsky *et al.* (2013) showed that climate velocity (the rate and direction of climate shift) can explain shifts in species using databases of centroids of biomass from survey observations in North American waters (e.g. the Bering Sea; Figure 2). Pinsky *et al.*'s results suggest that marine species can effectively follow preferred environmental conditions, but these migrations may re-organize marine communities because responses to changes in the environment vary by species. In addition to altering community structure, changes in depth (Dulvy *et al.*, 2008) or crossing over

international boundaries may alter the catchability of a species by a given nation's fleets (e.g. Pacific halibut; Clark and Hare, 2006).

Changes in the productivity of fish stocks appear common in many settings. Szuwalski *et al.* (2015) demonstrated changes in estimated average recruitment over time that was unrelated to spawning biomass occurred in over 60% of stocks in the RAM Legacy Stock Assessment Database (Ricard *et al.*, 2012) and sometimes these changes coincided with shifts in large-scale indices of environmental conditions (Figure 3). For example, recruitment for yellowtail rock fish in the Gulf of Alaska (Canadian stock) shifted up in the late 1970s at the same time winter sea surface temperature shifted upwards [Figure 3; shifts were identified using a breakpoint algorithm similar to Rodionov (2004)]. This shift coincided with a regime shift in the physical environment that reorganized the ecosystem (Anderson and Piatt, 1999). Production can be altered by changes in several population processes: changes in temperature can cause changes in growth (Hare, 2012), changes in phenology can produce changes in recruitment (Cushing, 1990), and changes in current patterns can influence prey availability for larvae and

juveniles and/or settlement in suitable nursery areas (Wilderbuer *et al.*, 2002). Natural mortality can be influenced by many variables (Walther *et al.*, 2002; Genner *et al.*, 2004). Changes in fish production exhibit decadal or multi-decadal patterns for many of the stocks in the RAM Legacy Databases, but the changes in productivity for stocks vulnerable to climate change may be "one-way trips".

"One-way trips" occur when a stock has been depleted to a small fraction of the estimated unfished biomass or its proxy and then it does not recover when fishing pressure is reduced. The causes of depletion may include overfishing or changes in environmental conditions influencing survival; the causes of failures to recovery include changes in trophic structure (Frank *et al.*, 2005), depensatory effects (Liermann and Hilborn, 2001), and changes in environmental conditions (Anderson and Piatt, 1999). Ten per cent of stocks that have estimates of recruitment, spawning biomass, and fishing mortality (22 of 225) in the RAM Database exhibited this pattern (Figure 4, bottom row for examples). Determining the cause of a one-way trip is difficult, but the changes in annual fish production are measurable (Bousquet *et al.*, 2014). There are a number of strategies used
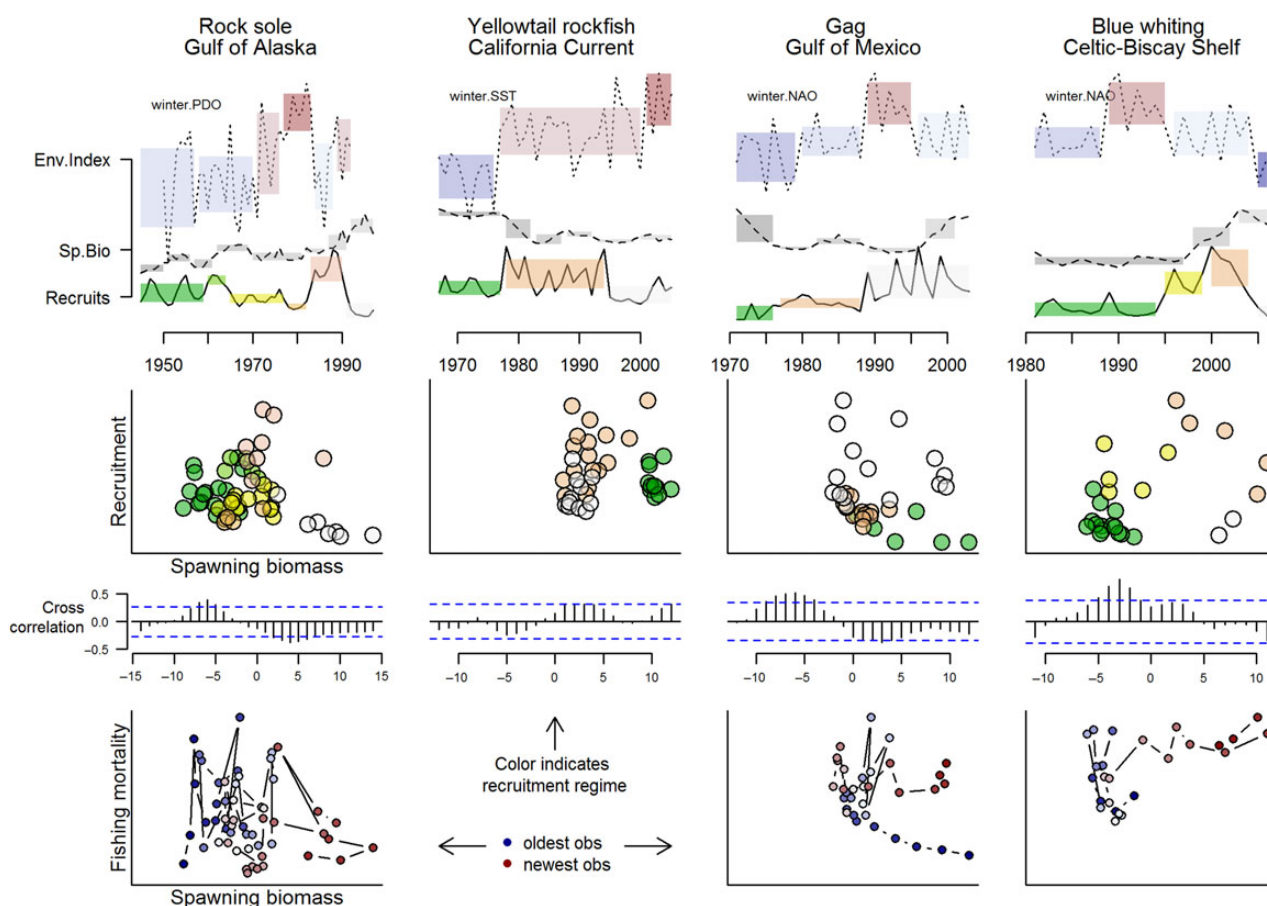

Figure 3. Fishery dynamics for four fisheries that display regime-like productivity in the RAM Legacy Stock Assessment Database. Labels are large marine ecosystem and common name (some large marine ecosystems have multiple stocks of a given common name). Top row shows time-series of an environmental index that is important in the large marine ecosystem in which the species occurs. Coloured boxes around the time-series are "regimes" identified by a breakpoint algorithm. Recruitment is plotted against spawning biomass in the second row, with the colour matching the "recruitment regime" from the top row—values range from 0 to the 1 for each quantity, scaled to the maximum observed. Cross correlation of spawning biomass and recruitment (third row) indicates the relative strength of the influence of spawning biomass on recruitment (zero lag correlation) and recruitment of spawning biomass (all negative lags are the influence of recruitment × years in the past on today's spawning biomass). Tracing a stock through the phase space of fishing mortality and spawning biomass (bottom row) over time shows the status of the stock today relative to the past. This figure is available in black and white in print and in colour at *ICES Journal of Marine Science* online.
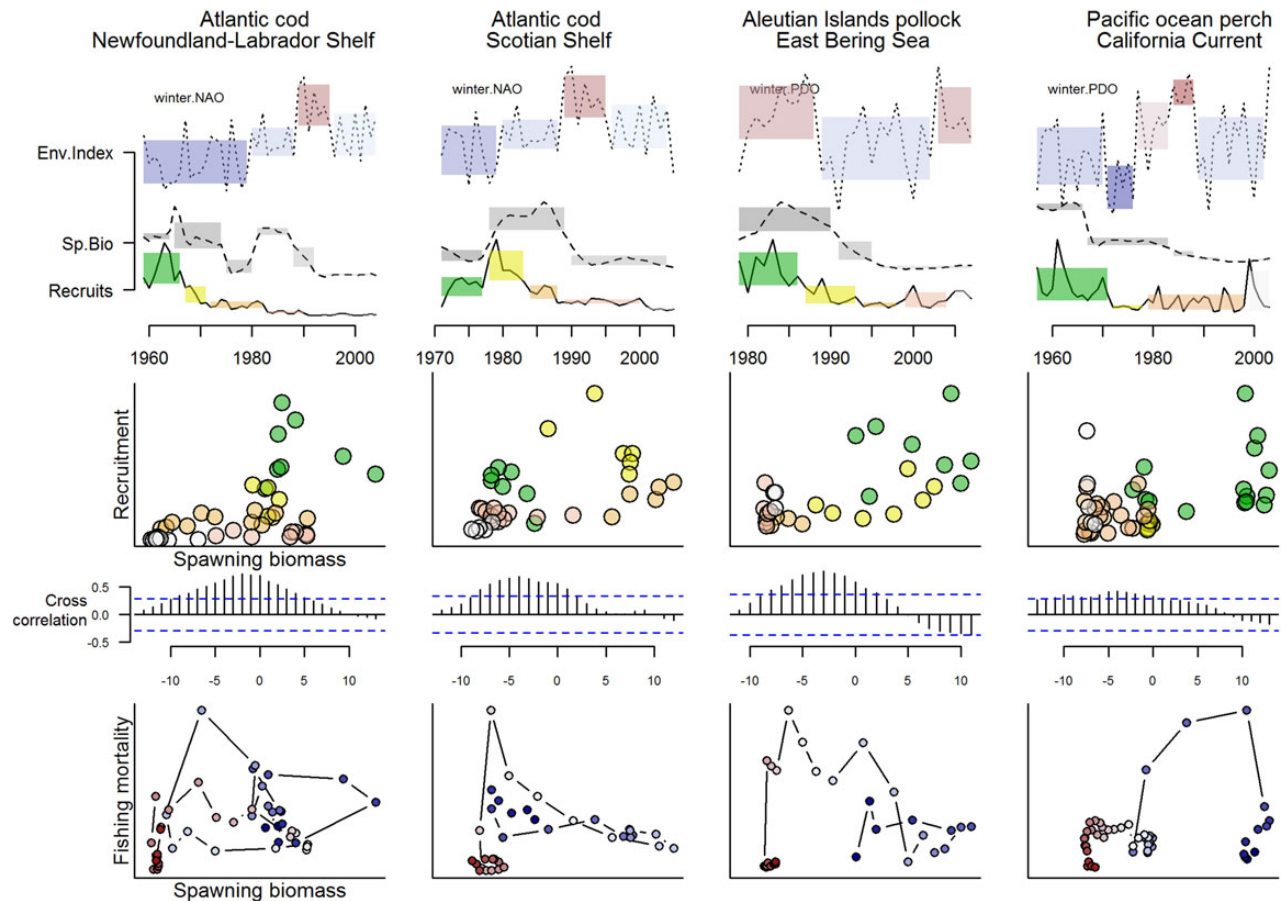
Figure 4. Fishery dynamics for four fisheries that display one-way trip behaviour. Same description as Figure 3. This figure is available in black and white in print and in colour at *ICES Journal of Marine Science* online.

to prevent depletion (e.g. setting a minimum stock size below which directed fishing is prohibited or decreasing fishing mortality as the stock decreases beneath the target spawning-stock biomass; Methot *et al.*, 2015). Strong constraints on fishing mortality may be imposed to rebuild depleted stocks (Cardinale *et al.*, 2013; Oremus *et al.*, 2014). These management actions may not only influence the target fisheries, but other fisheries for species that are incidentally caught with it (e.g. North Sea cod and discarding, Horwood *et al.*, 2006).

Assessment and reference points under non-stationary dynamics

Three key questions should be addressed to design sustainable harvest strategies under a changing climate: (a) how should the productivity of a fish population be projected into the future? (b) what level of precaution should be imposed? and (c) what metrics should be used to evaluate the performance of proposed management strategies? Stock assessments provide estimates of status, biomass, and parameters related to population processes which can be used to evaluate management. These parameter estimates can be used to project future population dynamics and calculate the reference points used in management. Non-stationary changes in population processes (e.g. environmentally driven growth or recruitment) present two problems for this process: retrospective

biases in estimated biomass and difficulties in calculating reference points.

A retrospective bias in a stock assessment is a "systematic inconsistency among a series of estimates of population size, or related assessment variables, based on increasing periods of data" (Mohn, 1999). Retrospective biases have been identified for a number of stock assessments (e.g. Butterworth, 1981; Sinclair *et al.*, 1991; Hamazaki and Zheng, 2012; Valero, 2012) and are thought to generally arise from contradictory data or unaccounted for variation over time in population processes (Parma, 1993). Retrospective bias has been observed in some statistical catch-at-age and length-based assessments when processes like natural mortality, growth, and selectivity vary over time in simulated populations (Hurtado-Ferro *et al.*, 2015). A variety of statistical methods have been developed to address retrospective biases, including allowing population processes to vary over time in the assessment model (e.g. Martell and Stewart, 2013; Thorson *et al.*, 2014).

Calculating biological reference points for management under projected time-varying environmental conditions is challenging because the stock must be projected into the future. Failing to account for time-varying processes results in biased biomass and fishing mortality-based reference points (Thorson *et al.*, 2015a). Several examples of evaluations of the performance of environmentally linked management strategies exist in the literature, but the results rarely lead to changes in current harvest strategies (Fogarty *et al.*, 2008; A'mar *et al.*, 2009; Hare *et al.*, 2010; Ianelli *et al.*, 2011;

Wilderbuer *et al.*, 2013). Szuwalski and Punt (2013) showed that changing management targets in a regime-based system can offer small increases in yield but they do so at an increased probability of overfishing. Even when switching strategies can improve long-term yields under regime-shift scenarios, determining the "true" underlying dynamics (i.e. whether or not the system is truly regime-based) can require extended periods of reduced exploitation (Simmonds and Keltz, 2007). Punt *et al.* (2014a, b) found that incorporating environmental forcing into the calculation of reference points improved outcomes when the mechanisms between the forcing and changes in population processes are well known; however, this level of understanding was rare in the case studies looked at by Punt *et al.* One of the potential shortcomings identified by Punt *et al.* (2014a, b) is that studies often consider temporary shifts in environmental conditions. The benefit of considering the influence of the environment on productivity will likely be much higher if non-stationary forcing (i.e. forcing that does not change back to the original state) is introduced.

These examples illustrate that the tools currently used to manage exploited populations have shortcomings when applied to stocks for which population processes are changing over time. If the fisheries management community is going to continue to follow the current approach to setting harvest controls (i.e. setting management targets for individual species based on reference fishing mortalities and biomasses), the issue of non-stationarity should be addressed.

An example

Some of the practical difficulties might be more easily understood with an example. Historically, the population of red king crab around the Pribilof Islands (PIRKC) was small (e.g. ∼100 000 crab; Szuwalski *et al.*, 2014; Figure 5). However, in the early 1980s, a large recruitment event occurred and the population increased by an order of magnitude by the early 1990s. A commercial fishery was opened in 1993 for PIRKC, but it was closed in 1998 after low survey estimates. The dynamics of PIRKC are an example of non-stationary dynamics—the productivity of the stock changed dramatically in the 1980s. Consequently, only years from 1984 forward are used in the calculation of the target biomass. Excluding the period of 1975−1983 on the basis that fails to reflect the current environmental conditions results in a ∼13% change in the calculated target biomass.

For single species management, the solution for PIRKC seems relatively straightforward: set reference points based upon the inferred productivity from the most recent "state" and harvest as
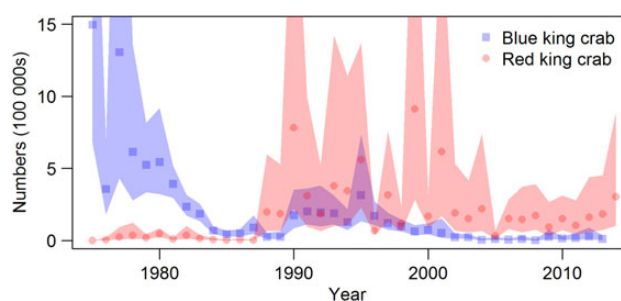

Figure 5. NMFS summer survey estimates of abundance (points) and confidence intervals (shaded areas) for blue king crab and red king crab around the Pribilof Islands. This figure is available in black and white in print and in colour at *ICES Journal of Marine Science* online.

usual. However, management for PIRKC becomes more complicated when other species in the area are considered. A population of blue king crab also exists around the Pribilof Islands (PIBKC) and had historically high abundances. Around the time of the increase in recruitment of red king crab into the area, blue king crab began a steep decline and have not recovered (Figure 5). PIBKC is considered overfished (Foy, 2013) and because PIBKC and PIRKC are caught in the same gear, a fishery for PIRKC has not been allowed to avoid impacting the PIBKC population. Currently, the total allowable catch calculated from stock assessments for PIRKC would have been 1359 t, which translates to over $21.7 million at market (Garber-Yonts and Lee, 2013).

The one way nature of climate change inevitably will present scientists and managers with challenging decisions similar to those in the PIRKC/PIBKC example. If the low population size of PIBKC is the "new normal" based on environmental conditions, then forgoing the potential yield from the PIRKC stock to rebuild the PIBKC stock to a population level that is no longer attainable might be a questionable management action. However, as resource stewards with many objectives, if there is a possibility that the PIBKC stock could rebound, protecting it from harvest until it does so may be a prudent course of action. This example illustrates the trade-offs that will likely emerge in the future. Scenarios of fish and fisher responses to future environmental conditions will be needed to evaluate how different management strategies perform relative to the ecological, economic, and social management goals when non-stationarity in productivity (or other population processes) becomes more common. Conflicts between "winners" and "losers" (in terms of changes in productivity) in mixed stock fisheries may become more common in the future, so frameworks must be developed and adopted in which decisions can be objectively made about changing management in response to changes in populations.

Looking forward

Frameworks for responding to non-stationarity induced by climate change must consider the same three points management currently grapples with (expectations for future dynamics used in the calculation of reference points, evaluation of the performance of management through stock assessments, and determining appropriate levels of precaution), but with special sensitivity to the changing nature of the system and potential conflicts between coincidentally caught species.

Confronting retrospective biases

Stock assessment plays a central role in management by providing estimates of biomass and fishing mortality which can then be compared with reference points to determine status. The retrospective biases that appear in biomass (and fishing mortality) estimates when population processes are non-stationary can be corrected for by allowing population processes to vary in the stock assessment (e.g. Schirripa *et al.*, 2009; Martell and Stewart, 2013). Determining which process is varying can be difficult with the data usually available for stock assessment and, to date, literature on the impacts of allowing the wrong process to vary over time within stock assessments on management targets is limited. So, two things need to be done to address the impacts of retrospective biases on management. First, simulation studies should be performed to understand the impact of misspecifying the time-varying process in stock assessments. If biases associated with misspecification in biomass or reference points are small, perhaps

allowing any process to vary that removes the retrospective bias is acceptable for management. Insensitivity of reference points to misspecified time-varying processes in stock assessment is an unlikely result, so cost-effective ways of collecting data to determine which processes are varying over time for commercial stocks must also be considered.

Projecting under uncertainty

If the process varying over time can be correctly identified and then allowed to vary in stock assessments, determining the character of the variability will be a key hurdle in projecting the stock to develop management targets. Calculating appropriate reference points would benefit from knowing if the process will revert to "normal" soon, revert to "normal" but only after the current cohort has died off, or if the current state is the "new normal". Mechanistic understanding of the causes of changes in population processes is a reasonable way to make decisions on the future trajectory of the stock when climate changes, but finding reliable relationships between environmental indices and recruitment (for example) is notoriously hard (Myers, 1998). Part of the poor performance of environment/recruitment relationships historically may come from the coarse scale at which "recruitment" is estimated. Recent advances in the spatial assessment of stocks may allow for exploring the relationship between environmental variables and populations processes at a finer scale (Thorson et al., 2015b) and possibly allow for the discovery of more reliable relationships. A reliable relationship between variables and populations processes can be coupled to the output of global climate models to project the stock under different future scenarios to evaluate candidate management strategies (e.g. Smith, 1994; A'mar et al., 2009; Ianelli et al., 2011; Szuwalski and Punt, 2013).

Trophic interactions also have the potential to influence population processes (e.g. natural mortality). Models of intermediate complexity for ecosystems that allow environmental effects on population dynamics and trophic interactions could be used to inform reference points (e.g. Plagányi et al., 2014; Punt et al., in press). Multispecies models (MSMs) are attractive because they formally address time-trends in mortality in a mechanistic manner (Holsman and Aydin, 2015). In MSMs, trophic interactions are governed by predator/prey relationships, which are often described by $N \times N$ matrices (where N is the number of species in the model). Diet data used to estimate these matrices are often limited and the variability in prey selection (with respect to the available prey and the life stage of an individual) is uncertain. Size-spectrum models sidestep some of the required information about diet by assuming prey selection is based on size rather than species (Blanchard et al., 2009, 2012), but still require spatial knowledge of the overlap of species (e.g. Blanchard et al., 2014). Using MSMs to identify potential future scenarios for natural mortality and then incorporating those future scenarios into management strategy evaluations may be a fruitful compromise between the two modelling paradigms (Spencer et al., this volume).

Estimates of quantities used in management provided from stock assessments have varying levels of uncertainty associated with them due to measurement error in the data, error in the processes modelled within the assessment, and model misspecification. These uncertainties can be (and often are) incorporated into the management advice provided based on assessment output. Model misspecification (e.g. selection of an inappropriate functional form for growth or mortality) contributes to uncertainty in management quantities (Buckland et al., 1997; Brodziak and Piner, 2010).

Model averaging has been used in the past for stock assessments (Brodziak and Piner, 2010), but averaging requires that the same data and likelihood functions are used in order for the output to be comparable. Many assessment techniques have been developed over the years which often have different data requirements (e.g. virtual population analysis and statistical catch-at-age, Quinn and Deriso, 1999). Stewart and Martell (2015) proposed a framework in which different modelling methods can be objectively compared for circumstances in which it may be desirable to compare the results for a range of models. Ensemble modelling is used extensively in the climate modelling world (Murphy et al., 2004) and may be a useful tool for projection of exploited fish populations (Ianelli et al., in press).

Fisheries managers need the capacity to ask "what if" in a quantitative way and then identify harvest control rules that can satisfy a range of often conflicting goals under a range of possible future scenarios (Groeneveld et al., in press). Criteria need to be developed to determine when and if a shift in population productivity should be declared [Klaer et al. (2015) begins this process]. Simulations can be used to thoroughly explore the costs, benefits, and risks of different management strategies under different future scenarios (e.g. Fulton et al., 2014). Harvest control rules robust to a changing environment have been searched for in many simulation studies (e.g. Kell et al., 2006; Deroba and Bence, 2008; Haltuch and Punt, 2011; Dickey-Collas et al., 2010; Punt, 2011). Constant fishing mortality strategies have been suggested to cope with climate change's influence on fished population (e.g. Walters and Parma, 1995), but if climate change influences processes other than recruitment, optimal fishing mortalities will change over time as well. If processes influenced by climate change cannot be identified and appropriately adjusted in projections, calculations of target fishing mortalities will be biased. Oftentimes, rules robust to changing climate will be less precise because they limit the amount of data that can be used to inform projection. For example, stocks managed by the North Pacific Fishery Management Council adjust the period over which data are used to calculate the reference points used to set allowable catches based on prevailing environmental conditions (Sustainable Fisheries Act of 1996). Recruitment estimates are available dating to the 1960s for some species, but only recruitment estimates from the late 1970s forward are used in calculation of proxies for reference biomasses because of an environmental shift (Mantua and Hare, 1999). Estimates of population parameters used to inform projections of the stock are less precise when using a subset of the available data, so increased precaution in setting allowable catches may be necessary given decreases in precision as we move into a more uncertain future.

Mechanistic understandings of exploited population dynamics are the best hope for accurately assessing and projecting responses when the environment changes (sensu Petitgas et al., 2013). Many modelling frameworks are available to attempt to describe mechanisms thought to influence the dynamics of fished stocks (e.g. Atlantis; e.g. Link et al., 2015), but some middle ground must be sought in terms of complexity (e.g. "models of intermediate complexity"; Plagányi et al., 2014). Developing mechanistic understandings of commercially exploited marine populations will require refinements in fisheries oceanography approaches (Hollowed et al., 2009; Bograd et al., 2014). Although the difficulties of predicting the impacts of climate change and appropriate management are couched in a fisheries context here, it is relevant in all disciplines in which the ability to "manage" a resource depends on the ability to predict the response of a system to a given disturbance (e.g. agriculture, finance, forestry). Moving forward, managers should focus

on predicting the dynamics of managed resources under climate change scenarios, characterizing the uncertainty around the potential climate scenarios and responses of resources to those scenarios (see Payne *et al.*, 2016), and then develop frameworks to make risk-based decisions among candidate management strategies, particularly when "winners" and "losers" of climate change must be co-managed.

Acknowledgements

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