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RESEARCH ARTICLE

Fishery management amidst spatially differentiated ecological-economic externalities

Linda Fernandez[1], Brooks Kaiser[2] and Melina Kourantidou[2,3]

[1]Department of Economics and Center for Environmental Studies, Virginia Commonwealth University, Richmond, VA, USA, [2]Department of Business and Sustainability, University of Southern Denmark, Esbjerg, Denmark and [3]CNRS, Univ Brest, Ifremer, IRD, EMR 6004, AMURE, IUEM, F-29280 Plouzane, France

Corresponding author: Linda Fernandez; Email: lmfernandez@vcu.edu

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Abstract

We study management decisions made jointly and independently by countries affected by an invasive species that is also a profitable fishery. The Red King Crab, introduced in Russian waters of the Barents Sea, spread into Norwegian waters. Management by Russia and Norway reflects differing markets and invasion damages. Our spatial dynamic bioeconomic model evaluates management of the crab and optimal game strategies integrating varied incentives from market prices, ecosystem values, and spatial connectivity. Our empirical application characterizes stock changes responding to different model components. This research shows economic and ecological trade-offs in Arctic waters with differing net benefits for sovereign stakeholders.

Keywords: applied game; Barents Sea; fisheries and invasive species; market and ecological values; red king crab

JEL Codes: D02; Q22; Q57

Introduction

Red King Crab (*Paralithodes camtschaticus*) (RKC) has been a valuable commercial species in the Bering Sea for a century (Dvoretsky and Dvoretsky, 2018; Otto and Stevens, 2014). Its high commercial value incentivized Russian scientists to introduce the species to the Russian Barents Sea multiple times over the 20th century with the intent to create a commercial fishery; establishment in the new region took hold in the 1960s (Kuzmin and Olsen, 1994, Orlov and Ivanov, 1978). The RKC has since spread and established in Norwegian waters of the Barents Sea (Pedersen et al., 2006) where it has been recognized as an invasive species (Jørgensen, 2013) and managed for both its commercial and invasive properties affecting communities and ecosystems (Oug et al. 2018; Sundet and Hoel, 2016).

The introduction, invasion, and profitability of RKC has evolved to where Russian and Norwegian sovereign interests in the shared stock have diverged, despite longstanding cooperation on other Barents Sea fisheries. Norway's assessments have attributed higher ecological costs to the RKC's presence (Kourantidou and Kaiser, 2019a), but also higher economic potential relative to other fisheries opportunities in the countries. We examine the impacts of this divergence with a game theoretic bioeconomic model of management decisions over space and time. We use the model to compare outcomes of historical versus recent management of the two countries. We study how changes in the joint and sovereign management of RKC shared stock spread in the Barents Sea have reflected disparities in sovereign assessments of ecological and economic benefits and damages from the species' arrival and spread, and the costs of these disparities' role in reducing cooperation.

Our research design involves developing a bioeconomic model that we simulate through forward iteration of each sovereign country's differential game strategy as incentivized by the distinct ecological and economic conditions with available spatio-temporal data. We compare myopic decisions without updated information on stock spread, market damage value, and nonmarket ecosystem value, to cases where such information is included. We include variation in open access and quota zones through different incentives for nonmarket values, harvest costs, market prices and stock dispersal with varied fisher participants and goals of balancing fishery and invasive species management.

Since the purposeful introduction, Russian management decisions have aligned with fishery profitability goals for the RKC while Norwegian management decisions have included efforts to balance fishery profitability with habitat conservation, particularly in service to other commercial species (Fiskeri-og Kystdepartement, 2007). We show how both an open access and a quota-limited zone may function in tandem in a fishery whose range is changing. Our analysis has implications for Arctic fisheries where marine invasive species exist amidst dual goals (fisheries development and habitat conservation), where range shifts for commercial species result from climate shifts, and where evolving economic and biological information can support policy for existing sovereign non-cooperative strategies.

Literature review

Invasive species may have market value through commercial trade as well as both market and nonmarket values from invasion impacts (Dalmazzone and Giaccaria, 2014; Springborn et al., 2011). This is the case for the RKC in the Barents Sea. The literature treats the trade-offs between commercial value gain and negative impacts from invasive species growth as static relationships without spatial dynamics.

With spatio-dynamic bioeconomic analysis of the RKC invasion in the Barents Sea, we add to the literature reviewed previously by Eiswerth et al. (2018) of economic analyses of invasive species that include time and space dimensions in management or policy with increased integration of ecological and economic asymmetries in a transboundary case.

The RKC invasion in the Barents Sea has interested economists and ecologists for at least a dozen years. Falk-Petersen et al. (2011) provide an early scientific overview of the invasion and its risks that is focused on potential benthic impacts and interactions with other fisheries. Falk-Petersen and Armstrong (2013) model the bioeconomics of RKC in a Norwegian fjord of the Barents Sea (Varangerfjorden) without transboundary biology or spatial relationships, nor explicit consideration of damages to benthic habitats. Our analysis expands bioeconomic modeling to include these important spatio-temporal factors. Kourantidou (2018) expands bioeconomic modeling of RKC that includes ecological damages throughout Norwegian waters but does not address transboundary concerns. Skonhoft and Kourantidou (2021) assess theoretically the gains from cooperation and trade-offs faced by two countries in the Barents Sea without seeking to empirically examine this theory. Kourantidou and Kaiser (2021) evaluate trade-offs between investing in managing the RKC invasion frontier vs. managing the quota-regulated fishery in Norway, again setting aside the international transboundary concerns. Kourantidou and Kaiser (2019a) explore research agendas driven by management goals and lack of cooperation within and outside the Barents region. They confirm that Norwegian and Russian assessments of ecosystem damages vary from one another and that Norwegian assessments in particular have changed over time.

Our bioeconomic analysis of spatially differentiated management in a binational application including market and nonmarket components of fisheries and invasive species trade-offs contributes to straddling stocks literature including Munro (1979) and Hannesson (1983). Several studies (Ekerhovd, 2010; Hannesson, 2006; Miller and Munro, 2004) assess the management of straddling stocks for other fisheries, without considering externalities such as those from an invasion.

Central to our empirical study is the impact of information about the RKC stock spatial spread for management in the Barents Sea, as found in stock assessments (Windsland et al., 2014; Michelsen et al., 2020) and impacts (Jørgensen and Spiridonov, 2013; Oug et al., 2011, 2018), joint IMR-PINRO annual trawl surveying through the Barents EcoSystem Survey (BESS), and CPUE data for the fishery. Information for public involvement in harvest of invasive species (i.e., in the open-access fishery) can help habitat conservation linked to biodiversity (Pasko and Goldberg, 2014). We compare RKC stock management with full information of quantified stock spread and values of damages to cases without full information for fishery and invasive species management. Our empirical analysis explores Nash solutions that are nonlinear in the invasive species stock, as an application of the theoretical analysis of Dockner and Long (1993). Our empirical context for simulation of fishery policy change and market incentives between Russia and Norway embodies what Dockner and Long (1993) theorized of the non-cooperative Nash equilibrium as a self-enforcing policy due to adaptive management when integrating full economic and environmental information.

Relevant to our work at a broader (non-crab specific) level, Sumaila (1997) informs our work with studies of strategic interactions in Barents Sea fisheries; Sanchirico et al. (2021) have theoretical spatial and dynamic solutions without empirical parameters for general connected fisheries areas; and Albers et al. (2010), Epanchin and Hastings (2010) and Kaiser and Burnett (2010) address invasive species management with spatial heterogeneity in terrestrial examples with negative impacts of invasive species

Ehtamo and Hamalainon (1989) discuss the need to recognize asymmetric incentives across countries with different fisheries and pollution impacts in the Baltic Sea; this mirrors the asymmetries in the Barents RKC case. Fernandez (2006, 2007) addresses asymmetric incentives involving fisheries losses and pharmaceutical benefits between countries that accidentally introduced invasive species, providing an example of differential biodiversity costs and benefits akin to those in Norway and Russia. Pasko and Goldberg (2014) investigate market-driven harvest targeting a conservation goal, including bounties and commercial activities. They find that open access harvesting of an invasive species can help achieve conservation goals if properly managed to avoid incentives to expand the target species population further. Kourantidou (2018) discusses the efficiency of Norwegian use of subsidies to reduce RKC spread in Norway's open-access zone.

Background of the fisheries

The Barents RKC distribution is divided across three zones, identified from east to west in Figure 1 as *A*, the Russian Zone; *B*, the eastern Norwegian zone; and *C*, the western Norwegian, invasion frontier zone. In early years of the RKC's presence in the Barents Sea, Norway and Russia agreed under the Joint Norwegian Russian Fisheries Commission (JointFish) to not fish for the RKC but instead jointly manage it through the 1978 Grey Zone Agreement. Cooperation between Russia and Norway on Total Allowable Catch (TAC) decisions occurred under this agreement through 2006. The countries ran experimental fisheries from 1994 to 2001 with the same quota for each country (Dvoretsky and Dvoretsky, 2018).

Bycatch costs in Norwegian cod fisheries (Sundet and Hjelset, 2002) were initially managed cooperatively, with a Norwegian RKC fishery limited to those seeking bycatch damage reparations. A gradual understanding of RKC invasion impacts to benthic ecosystems (Jørgensen and Spiridonov, 2013; Oug et al., 2011; Pavlova, 2021) as well as predation upon other commercially valuable species such as the capelin and lumpsucker (Mikkelsen and Pedersen, 2012; Mikkelsen, 2013), prompted a change in management for invasive species control. Russia and Norway differed in their perception of damage from the invasive RKC (Anisimova et al., 2005; Dvoretsky and Dvoretsky, 2018; Kourantidou and Kaiser, 2019a; Tsygarova et al., 2015) and thus in their management objectives for the fishery. Norway embarked on a commercial fishery in coastal waters in 2002, Russia began its commercial fishery offshore in 2004. The opening of the fishery was believed to serve as a barrier to the continued westward expansion of the crab's habitat. In 2005, Norway and Russia agreed to establish a western boundary at 26ºE, to allow for an open-access fishery on the invasion frontier (west of 26ºE) (Sundet and Hoel, 2016). The invasion's westward spread into Norway (Zone *B*) was initially managed jointly by Norwegian and Russian authorities, through quotas that aimed to alleviate the burden on coastal cod fishers who were experiencing cod bycatches. Commercial interest in the red king crab fishery led to opening the fishery for commercial exploitation and partly alleviate those bycatch losses. Norway expanded quota access beyond bycatch reparations, to anyone residing in Eastern Finnmark. Despite a signed agreement for joint research efforts via a three-year research program (2005–2007) and the agreements in force since 1993 (Joint Norwegian Russian Fisheries Commission, 2005), Russia established quota limits for the Russian zone unilaterally without providing previous notice to Norway (Øseth, 2008). In 2006, the two countries agreed to shift from a joint to a national management of the species separately within their respective domains (Eriksen, 2008). Onwards, the Norwegian zones (*B* and *C*) have spatially differentiated management plans independent of Russian political consultation.

Post 2006, the differences in objectives have led to each country operating separately. Russia manages Zone *A* (Figure 1) as a quota-regulated offshore RKC fishery, while Norway has institutionalized split management of its coastal fishery, with Zone *B* (Figure 1), the area east of 26°E and south of 71°30'N (*B*), has been managed by non-transferable individual vessel quotas for sustaining a long-term fishery, and Zone *C* (Figure 1) as an open-access western, frontier zone aimed at minimizing the spread of the RKC west of the 26°E boundary (Fiskeri-og Kystdepartement, 2007). Thus, the RKC exists today in this two-country transboundary setting with spatially variable economic benefits and ecological damages (Kourantidou and Kaiser, 2019a; Sundet, 2014). This is a significant coordination failure placing the sovereign fishery management regimes at odds with JointFish cooperation mechanisms governing other shared fish stocks for the first time since the 1970s.

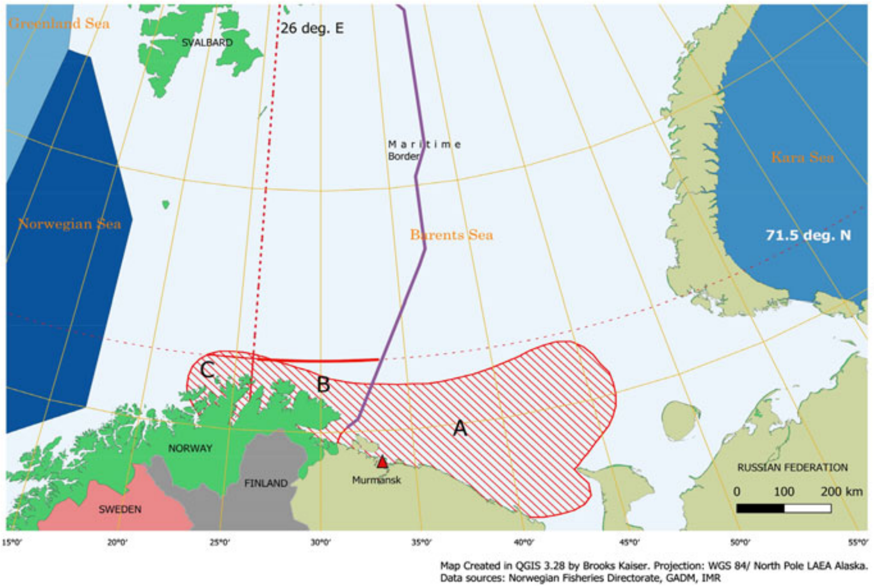

Figure 1. Map of the Barents Sea with Red King Crab fishery zones across Russia (A) and Norway (B east of 26°E known as the quota-regulated area and C west of the 26°E known as the open-access area). The red triangle in area A depicts the RKC introduction point near Murmansk.

Historical management lacked data for understanding RKC spatial connectivity, and focused on a limited spatial scale, i.e., individual fjords (Michelsen et al. 2020; Oug et al. 2011, 2018; Sundet, 2014). Annual stock assessments by both countries through the Norwegian Institute for Marine Research (IMR) and Russia's Polar Research Institute of Marine Fisheries and Oceanography (PINRO) have generated information on spread and spatial connectivity of the RKC distribution (see Figure 1) (Joint Fish, 2004). For example, a tagging study from 1994 to 2011 quantified adult spatial connectivity, finding predominantly westward expansion from Russian waters into and across Norwegian waters (Windsland et al. 2014). The red triangle in Figure 1 depicts the RKC introduction point near Murmansk. Ongoing study of RKC dispersal seeks to uncover the mechanisms for the westward spread (Hønneland et al., 2020; Windsland et al., 2014).

The annual quotas and total allowable catch (TAC) in Norway are decided by the Norwegian Ministry of Trade, Industry and Fisheries (previously Ministry of and Coastal Affairs), with advice from the Directorate of Fisheries which is based on annual stock assessments for the quota-regulated area provided by the IMR, see e.g., Hvingel and Hjelset (2022). Sundet and Hoel (2016) emphasize Norway's commitment to the Convention on Biological Diversity (CBD) which is to prevent invasive species spread into international waters[1]. It remains unclear whether Norway's strategy to pursue a long-term fishery

[1]Projections indicate that the RKC may expand to the current- and temperature-delineated border dividing the Barents and Norwegian Seas, moving South to Trøndelag in 30 years, past the current location of longitude 69°N near Tromsø (Saetva, 2019, Saveson, 2019, Hvingel et al, 2022). Still, the expected northern movement of RKC with coastal current larval advection to Bear Island studied by Pedersen et al (2006) (projected to arrive by 2025) and to Svalbard by 2035, is within Norwegian sovereign waters (Saetva,

alongside meeting CBD goals for the invasive RKC with a split management scheme can achieve optimal outcomes, given the absence of explicit considerations of the spatial spread and continuing expansion of the invasion (Michelsen et al., 2020). In this paper we investigate the question empirically.

The differentiation of management regimes across space and time has also affected the distribution of commercial gains and political pressures on overall quota levels. As mentioned, initial Norwegian RKC quotas were awarded as bycatch compensation, followed by residency-based quota access that increased fishing pressures. Later, as the value of the RKC fishery grew, political pressures from incumbent quota holders, evidenced in annual stakeholder meetings, prompted more entry restrictions for commercial fishers by requiring minimum fishery revenues from other fisheries (i.e., cod), to avoid opportunists and to limit benefits primarily to coastal residents and those affected by the RKC invasion (Fiskeridirektoratet, 2017). Simultaneously, political pressure from potential entrants exists to shift the quota zone further west of the 26° East border, which would result in allowing access to more fishers to an expanded quota-regulated area with increased profitability potential.

These Barents Sea RKC fisheries are evolving in a globally traded commodities market. Alaskan and Far East Russian fisheries dominate crab supply to the market. World market prices for both live and frozen RKC drive incentives for its harvest. Live RKC commands a higher world price than frozen RKC. Russia and Norway respond to market prices and demand in different ways, reflecting differing access to supply and final markets. Distant markets for RKC from the Barents Sea affect the net benefits of harvest, particularly as live harvest commands higher prices but also greater logistical costs. Voldnes et al. (2020) describe marketing and value chain challenges for Norway aiming for year-round supply of perishable live crab to distant markets such as South Korea, as Norwegian quantity transported live has grown from zero initially to ~70% of harvested RKC in recent years. Lorentzen et al. (2018) reinforce the focus on live RKC for Norway, which requires costly vigilance to avoid mortality before sale.

Materials and methods with theory

The expansion of the RKC invasive species in the Barents Sea is assumed to be unilateral in direction from Zone *A* through Zone *B* to Zone *C*, as Figure 2 illustrates conceptually.

We divide the invasion space according to both ecological and economic heterogeneity with recognition of the spatial link between the zones, with zone *B* receiving crabs from *A* and sourcing them to *C*, noted in the middle of Figure 2. Differences in invasion stage, source of invasion and impacts are noted in boxes next to each of three zone labels. Zone *A* is fully invaded while Zone *B* is experiencing continued spread and/or increasing volumes. This general context applies to Russia and Norway for our empirical analysis, as shown with the same zones in both Figures 1 and 2.

Cost of harvest

Harvest costs for RKC may differ spatially depending on harvesting technology or other asymmetries across countries. Costs influencing the final product may also depend on technological investment related to processing costs/handling time. Thus, investments in shipboard technology (e.g., flash-frozen capacity) can lead to lower marginal costs, but the

2019, Saveson, 2019). There is no evidence to date that climate change has significantly affected distribution of RKC; this may be an interesting avenue for future research to follow concerns expressed in Christiansen et al. (2015).

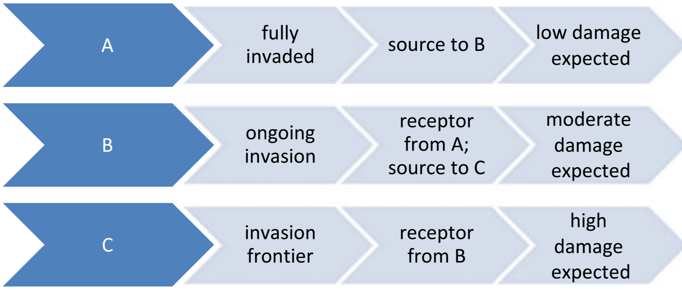

Figure 2. Invasion process and expectations over damages.

associated changes in the quality result in lower prices compared to live crab. Investment in onshore capacity to help supply live crab to market at a higher price increases marginal costs of the harvest. For Russia, lower transport costs as well as the higher market price for live crab supplies from the Russian Far East to Asian and U.S. markets make the frozen crab from the Barents Sea less competitive. Between 20 and 30 vessels partake in the Russian Barents Sea fisheries with offshore onboard processing, and no significant investment in live crab has been made to date (Hønneland et al., 2020; Urner Barry, 2020). Several of the vessels and quota are owned by companies that also participate in the Far East; the Russian crab quota is highly concentrated (Kalinin and Vershinin, 2020). This reinforces our understanding of Russian Barents Sea crab as a reserve population with lower marginal returns compared to Far Eastern supplies; this hinders substantial Russian investment in live crab production. For Norway, there is a capacity constraint on live crab stemming from logistics for delivering live product safely and rapidly to distant markets (Hodges, 2020). It is not a smooth transition to shift from live to frozen with current limits to capacity that COVID-19 exacerbated (Hodges, 2020).

Harvest technology regulation determines the cost function from one of two possibilities (quantity limits on vessels or harvest). Regulations that impose, e.g., vessel size limit restrictions (in Norwegian waters) limit efficiency and result in higher marginal costs. The coastal catch from these vessels is processed onshore rather than aboard ship offshore (as occurs in Russia), divided between high-price live sales and low-price (frozen) sales, and is a function of the investment in onshore infrastructure development for live supply capacity and mobile receiving stations.

We assume that onshore infrastructure investment for live output is exogenously fixed and sets a capacity constraint. We further assume that this infrastructure is zero for Zone A (based on empirical evidence), so that all harvest in Zone A will use the larger scale harvest technology offshore at lower marginal cost.

Management may set quota regulations on harvest and harvest technology limitations which affect the direct net benefits of harvest. The managers select harvest and technology that determine not only the level of extraction in their waters but also the quality of the product delivered to market. For example, flash-frozen production has higher fixed costs but some economy of scale by volume. Flash-frozen RKC sells for a lower price compared to fresh legal size male RKC. Technology needs raise the cost of harvest and we assume this has induced onshore investment M (exogenous to the managers' choices) that allows for higher quality (live) harvest (h_H) that can be sold at a higher price, p_H. Processing capacities for high quality output are set at $\overline{h}_{Hi}$ respectively, where i refers to the relevant zone.

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For this analysis, we assume that onshore investments exist only in Zone B, so that $M_B > 0$. RKC production from the Barents Sea Zone A consists of Russian managers setting technology rules that allow cheap on-board processing, while Norway restricts harvest technology to small coastal vessels. In Zone C, lower and less certain crab densities have reduced catch and the benefits of investing in live market access and high price. Thus $\overline{h}_{HA}=\overline{h}_{HC} = 0$ and price in these zones is low, p_L.

Population dynamics

Population dynamics of the stock S_i in each zone (i) are modeled as a unidirectional Fisher/KPP diffusion (dispersal) model with logistic internal growth (Shigesada and Kawasaki, 1997) to capture spatial transfer from source zone A to receptor zone B, and subsequently from source B to receptor C. The model has a uniform intrinsic growth rate r based on its assessment as a single stock (Hønneland et al., 2020), and each zone has a carrying capacity, $K_{\rm i}$. The harvest, h_{n_i}, may be split amongst N_i participants in the zone's fishery, so that each region follows a dynamic net growth equation (1.1) for change in the RKC stock S_i between time periods of:

$$\dot{S}_i = rS_i\left(1 - \frac{S_i}{K_i}\right) - \mu_i(S_i) + \mu_j\left(S_j\right) - h_i, \mathit{for}\; i = A, B, C, j = A, B \tag{1.1}$$

where $\mu_i(S_i)$ is the stock-dependent diffusion transfer (of unit distance) out of zone i (unidirectional, indexed across A,B,C), $\mu_j\left(S_j\right)$ is the stock-dependent spatial transfer (unidirectional, e.g., from "east across the border," indexed across A,B) into zone i, and h_i is the harvest in zone i, in time t. We assume $\mu_i^{'}(S_i) \geq 0, \mu_i(0) = 0, \mu_i(K_i) = \bar{\mu}_i$. We assume a simplified linear unilateral spatial transfer out of zone i, $\mu_i(S_i) = \mu_i \cdot S_i$ as measured from Russian and Norwegian studies (Hønneland et al., 2020; Pinchukov, 2009; Windsland et al., 2014). The measure of spatial transfer between areas includes feedback effects over time (and space) between zones with S representing all biomass instead of age and density dependent dispersal given there are ongoing studies to sort out actual mechanisms for dispersal of RKC (Aune et al., 2022; Dvoretsky and Dvoretsky, 2022; Windsland et al., 2014). We also assume harvesting RKC does not cause RKC to flee one zone for another. Norwegian fjord-scale models note some spread of RKC between fjords (Michelsen et al., 2020). The total harvest in a zone,h_i, is the sum of individual shares to fishers.

Zone A is the source of the invasion, so that $\mu_j\left(S_j\right) = 0$ for $i = A$. Saetva (2019) provides a forecast for RKC up to 2050 that includes stock movement via coastal current larval advection (Pedersen et al., 2006), and adult population walking to continue expansion in Zone C to the border of the Norwegian Sea. The stocks are considered separate, with different carrying capacity among the three zones, due to spatially varied topographical conditions. For example, Zone A consists of offshore open sea waters for Russia's harvest while Zones B and C consist of coastal fjord waters for Norway's harvest.

In Zone A, the resource stock S_A is considered beneficial, without acknowledging invasion damages tied to purposeful introduction (McBride et al., 2016). The population has fully invaded, and approaches carrying capacity (K_A) without harvesting. Marginal damages are expected to be approximately zero as benthic habitat has already changed from the established RKC (Anisimova et al., 2005; Zakharov, 2016). Baseline conditions for damages are not extensively documented, with only a couple scientific studies including Anisimova et al. (2005) and Zakharov (2016). Kourantidou and Kaiser (2019a, 2019b) note management authorities do not exhibit concern.

Zones B and C differ in the benthic habitat and in the infrastructure to support high quality harvests. In Zones B and C, there is ongoing change in net impacts of the invasive crab, and the zones are occupied by stocks S_B and S_C respectively (indexed by i). There are ongoing ecological damages to other commercially valuable species and benthic biodiversity due to predation by the invasive species, which we specify as a function of stock: $D_{Fi}(S_i)$, with subscript F for fisheries bycatch and predation damages, are non-decreasing in stock ($D_{Fi}{}'(S_i) \geq 0$).

Bycatch damages for the introduced species are understood as they tie directly to commercial fishery values. However, baseline benthic species values are not well measured. We link empirical nonmarket values on spatial benthic conservation, where spatial benthic value $V_i \geq 0$, has an expected value with variance over the public nonmarket values for each zone as:

$$dV_i = \nu_i dt + \varsigma_i dz, \tag{1.2}$$

The biodiversity value for any one species of the benthic habitat at risk is unknown. The total biodiversity value for a zone is subject to change across time as the composition may change due to the invasion or other ecosystem processes, as may scientific information regarding the role of the biodiversity and/or the relative and diverse uses of the benthic space for, e.g., fisheries or hydrocarbon exploration (Aanesen et al., 2015; Kourantidou and Kaiser, 2019a). The benthic habitat for a zone at time t has an expected nonmarket value of marginal benefit to the country's population[2] ν_i along with $\varsigma_i \sim \mathrm{N}(0, \sigma_i^2)$ as a variance component of value, normally distributed across space with mean zero and zone-specific variance σ_i^2. The standard deviation of the volatility is σ_i and dz represents the normally distributed increment of the Weiner process. The benthic habitat values and variances may differ across zones due to different composition of benthic habitat and/or different information or uses of benthic species. Our applied analysis ties equation (1.2) to known empirical measures of Norwegian public values included for deterministic representation in the applied optimization instead of stochastic.

The total stock-dependent damage function from the stock of the introduced species in a zone is then:

$$D(S_i) = D_{Fi}(S_i) + \nu_i dt + \varsigma_i dz \tag{1.3}$$

where the first term of the right-hand side of equation (1.3) consists of the known damages to commercial fisheries from the invasive crab. The second and third terms of the right-hand side of equation (1.3) show the value change and variance of benthic habitat value, which we link to empirical evidence from Norwegian contingent valuation on protecting benthic habitat space.

Management

We model and simulate sovereign management by zone for Russia and Norway to compare to historical management. Until 2006, Russia and Norway's historical coordinated management did not include joint economic optimization with revenue sharing. Instead, the historical management consisted of both countries agreeing on quantity restrictions not derived from joint economic optimization. The limited scientific knowledge regarding RKC stock spatial dynamics during the coordination period sets the stage for comparing with cases including updated information. We explore variation in management through the impact of information linking zones by stock spatial transfer as well as values tied to dual goals of invasive species control and fisheries in the following derivations of sovereign zone management optimization.

[2]These existence values are global public goods. The manager is not assumed responsible to include values outside the country in managing social welfare. We do include national values in our empirical analysis.

We compare myopic management (without updated information) with Nash non-cooperation that includes the updated information on stock spatial transfer, etc. We illustrate differences in biomass changes for optimally managed quota fisheries zone and static open access in the extreme. The derivations include shadow values that link two different management zones for Norway as a means to explore the role of information affecting incentives for public awareness to induce participation in open access to help with the two potentially competing goals of invasive species management and profitable fisheries.

We present the model in a non-cooperative sovereign context first and then describe the limited cooperation for Russia and Norway.

A fishery manager for a zone selects harvest restrictions (quotas) to maximize social welfare over time in the zone, given available technology, exogenous processing capacity and damages for output in the zone. The zone's dynamic problem for a fishery manager maximizing discounted net benefits is:

$$\delta W_i(S_i) = \underset{h_{Hi},h_{Li}}{Max} \int_t^{\infty} e^{-\delta t}\big(p_H h_{Hi}(M_i) + p_L h_{Li} - C_{Hi}(S_i)h_{Hi} - C_{Li}(S_i)h_{Li} - D_i(S_i)\big) \tag{1.4}$$

subject to stock growth indicated by equation (1.1), where p_L is the world price available to the lower quality (frozen) output, p_H is the world price available to the higher quality (fresh) output, $C_i(S_i)$ is the stock-dependent marginal cost of harvest, and $D_i(S_i)$ are the stock-dependent net damages to bycatch fisheries along with spatially varied habitat value impacts V_i by zone. $h_{Hi} \in \left[0, \bar{h}_{Hi}\right]$ where $\bar{h}_{Hi}$ is the maximum harvest that can be sold at the higher price due to capacity constraints determined by investment M_i. The discount rate is δ, W is the value function of the optimization and $\frac{\partial W_i}{\partial S_i}$ the partial derivative of the value function with respect to the stock variable is represented by λ_i the shadow value of the constraint in subsequent equations past (1.5).

The Hamilton-Jacobi-Bellman (HJB) equation for each manager is then:

$$\begin{aligned}\delta W_i(S_i) = H_i = \max_{h_{Hi},h_{Li}}\big(p_H h_{Hi}(M_i) + p_L h_{Li} - C_i(S_i)h_{Hi} - C_i(S_i)h_{Li} - D_i(S_i) - \frac{1}{2}\sigma_i^2 V_i^2 \\ - \frac{\partial W_i}{\partial S_i}\left(rS_i\left(1 - \frac{S_i}{K_i}\right) - \mu_i S_i + \mu_j S_j - h_i\right).\end{aligned} \tag{1.5}$$

We will now focus on each country beyond the general form.

In the case of Russia as Zone A, with price for frozen crab, p_L and no anticipated damages, the Hamiltonian H in equation (1.5) with i=A, becomes simply:

$$H_A = p_L h_{LA} - C_A(S_A)h_{LA} + \lambda_A(rS_A(1 - \frac{S_A}{K_A}) - \mu_A S_A - h_{LA}), \tag{1.6}$$

as $M_A = 0$; and lowest cost frozen technology is used.

The First Order Necessary Conditions are then:

$$\frac{\partial H_A}{\partial h_{LA}} = (p_L - C_A(S_A))e^{-\delta t} - \lambda_A = 0 \tag{1.7}$$

$$\dot{\lambda_A} = -\frac{\partial H_A}{\partial S_A} = e^{-\delta t}C_{A,S_A} - \lambda_A(r - \frac{2rS_A}{K_A} - \mu_A) \tag{1.8}$$

$$\frac{\partial H}{\partial \lambda_A} = \dot{S}_A \tag{1.9}$$

Taking the time derivative of equation (1.7)

$$\dot{\lambda_A} = -\delta(p_L - C_A(S_A))e^{-\delta t} - e^{-\delta t}C_{A,S_A}\dot{S_A} \tag{1.10}$$

And substituting for $\dot{\lambda_A}$ and λ_A,

$$\delta\big(p_L - C_A(S_A)\big) - \left(r - \frac{2rS_A}{K_A} - \mu_A\right)\big(p_L - C_A(S_A)\big) = -C_{A,S_A}\big(\dot{S_A} + 1\big) \tag{1.11}$$

$$\delta - \left(r - \frac{2rS_A}{K_A} - \mu_A\right) = \frac{-C_{A,S_A}\big(\dot{S_A} + 1\big)}{\big(p_L - C_A(S_A)\big)} \tag{1.12}$$

$$\delta = \left(r - \frac{2rS_A}{K_A} - \mu_A\right) - \frac{C_{A,S_A}}{\big(p_L - C_A(S_A)\big)} \tag{1.13}$$

Equation (1.13) is the fundamental equation of renewable resource economics which can be rearranged as in equation (1.14) to describe the optimal steady state fishery where Russian fishers choose between letting the RKC stock grow as natural capital appreciating or cashing in through harvest with revenues appreciating by the interest rate:

$$\left(r - \frac{2rS_A}{K_A} - \mu_A\right)\big(p_L - C_A(S_A)\big) - C_{A,S_A} = \delta\big(p_L - C_A(S_A)\big) \tag{1.14}$$

The return from retaining the last increment of the fish stock has two components: the value of increased RKC stock growth and the ability of the stock to reduce costs. These two benefits of holding the marginal unit of a stock are compared to the opportunity cost (return on the net of marginal revenue minus marginal cost invested after cashing in from harvest).

Rearranging equation (1.14) to solve for S_A implies comparative statics where decline of steady state stock results from increased spatial transfer out of Zone A, increased price for frozen RKC and decreased cost as incentives to harvest. Subsequently, the empirical section depicts Russian RKC from measures in the Barents Sea, including the spatial transfer μ_A that was estimated after fisheries management started. This measure is key for gauging updating between individual zones that are connected. The numerical solution is necessary with the empirical functions to move beyond general functional form steady state from the First Order Necessary Conditions above for the determination of how information on zone connectivity influences optimized stock, S_A:

$$S_A = \frac{\left(r - \delta - \mu_A - \frac{C_{A,S_A}}{(p_L - C_A(S_A))}\right)K_A}{2r}$$

In Norwegian coastal waters, if we treat Zones B and C the same except for price difference where Zone C has undersized crabs fetching only the low price, p_L, the Hamiltonian (1.5) becomes

$$\begin{aligned} H_{BC} = {} & \big(p_H h_{HB}(M_B) + p_L(h_{LC}) - C_B(S_B)h_{HB} - C_C(S_C)h_{LC} - D_B(S_B) - D_C(S_C)\big) \\ & + \lambda_B\left(\left(rS_B\left(1 - \frac{S_B}{K_B}\right) - \mu_B S_B + \mu_A S_A - h_{HB}\right)\right) \\ & + \lambda_C\left(rS_C\left(1 - \frac{S_C}{K_C}\right) - \mu_C S_C + \mu_B S_B - h_{LC}\right) \end{aligned} \tag{1.15}$$

We assume that M_B is infrastructure available for live RKC to be supplied to the world market with the focus on year-round harvest. The components relating to Zone C (shadow value and damages in equations (1.17), (1.18), and (1.19)) will become a basis of modifying open access by these factors in our empirical application to investigate effects on the stock. After optimization in Zone B, Zone C's open-access conditions are presented.

$$\frac{\partial H_{BC}}{\partial h_{HB}} = (p_H - C_B(S_B))e^{-\delta t} - \lambda_B = 0 \tag{1.16}$$

$$\frac{\partial H_{BC}}{\partial h_{LC}} = (p_L - C_C(S_C))e^{-\delta t} - \lambda_C = 0 \tag{1.17}$$

$$\dot{\lambda_B} = -\frac{\partial H_{BC}}{\partial S_B} = e^{-\delta t}C_{B,S_B} + e^{-\delta t}D_{B,S_B} - \lambda_B\left(r - \frac{2rS_B}{K_B} - \mu_B\right) - \lambda_C\mu_B \tag{1.18}$$

$$\dot{\lambda_C} = -\frac{\partial H_{BC}}{\partial S_C} = e^{-\delta t}C_{C,S_C} + e^{-\delta t}D_{C,S_C} - \lambda_C\left(r - \frac{2rS_C}{K_C} - \mu_C\right) \tag{1.19}$$

$$\frac{\partial H_{BC}}{\partial \lambda_B} = \dot{S}_B \tag{1.20}$$

$$\frac{\partial H_{BC}}{\partial \lambda_C} = \dot{S}_C \tag{1.21}$$

Taking the time derivative of equations (1.16) and (1.17),

$$\dot{\lambda_B} = -\delta(p_H - C_B(S_B))e^{-\delta t} - e^{-\delta t}C_{B,S_B}\dot{S_B} \tag{1.22}$$

$$\dot{\lambda_C} = -\delta(p_L - C_C(S_C))e^{-\delta t} - e^{-\delta t}C_{C,S_C}\dot{S_C} \tag{1.23}$$

Substituting for λ_B and $\dot{\lambda}_B$

$$\begin{gathered} -\delta\big(p_H - C_B(S_B)\big)e^{-\delta t} - e^{-\delta t}C_{B,S_B}\dot{S_B} = e^{-\delta t}C_{B,S_B} + e^{-\delta t}D_{B,S_B} - \\ \big(p_H - C_B(S_B)\big)e^{-\delta t}\left(r - \frac{2rS_B}{K_B} - \mu_B\right) - \lambda_C\mu_B \end{gathered} \tag{1.24}$$

$$-e^{-\delta t}\big(C_{B,S_B}\big(\dot{S_B} + 1\big) - D_{B,S_B}\big) + \lambda_C\mu_B = \left(\delta - r - \frac{2rS_B}{K_B} - \mu_B\right)\big(p_H - C_B(S_B)\big)e^{-\delta t} \tag{1.25}$$

$$\left(r - \frac{2rS_B}{K_B} - \mu_B\right) + \frac{-C_{B,S_B}(\dot{S_B} + 1) - D_{B,S_B} + e^{\delta t}\lambda_C\mu_B}{\big(p_H - C_B(S_B)\big)} = \delta \tag{1.26}$$

$$S_B = K_B\left(r - \delta - \mu_B + \frac{-C_{B,S_B}(\dot{S_B} + 1) - D_{B,S_B} + e^{\delta t}\lambda_C\mu_B}{P_H - C_B(S_B)}\right)/2r \tag{1.27}$$

Zone B may follow a modified golden rule of renewable resources (equation (1.26)) where the sum of marginal harvesting costs and marginal damages includes the shadow value of Zone C linked through Zone B stock spatial transfer. In this way, we explore variation in single zone myopia to consider internalized spatial dependence between zones.

The higher the harvesting benefits, the lower the optimal stock. In the steady state for the optimally managed quota, marginal benefits minus marginal damage costs and other costs equal fishery rents minus opportunity costs from the spatial transfer to Zone C (in equation (1.27)). In the individual vessel quota Zone B, Norway pursues Maximum Economic Yield (MEY) for RKC. Our empirical analysis will assess management varied by spatial transfer and values in (μ_A) and out (μ_B) of Zone B as well as damages to explore changes in stock beyond comparative statics from equation (1.27). The comparative statics show an increase in steady state stock with increased intrinsic rate of growth and alternatively, declining steady state stock with increased diffusion transfer out of Zone B, increased price for RKC, decreased cost and increased damages.

Zone C has open access with total harvest h_C involving stock, S_C, effort, E, and catchability, q, of fishing by recreational and other fishers. We also explore variation in Zone C comparing static open access where fishers are not informed of stock with cases of management including stock information (spatial transfer and location) as a means of incentivizing fishers in invasive species disruption as well as fishing where the zones are jointly managed. Equation (1.28) shows how all fishers impact stock when equating total costs to total revenues of pure open access with h_C represented on the left-hand side in parentheses by a Schaefer production function qS_CE, combining recreational and commercial fishers in area C. Costs as a function of stock link to available costs for Norway among various fishers without a survey of recreational fishers' explicit costs. The static context of open access with rent dissipation where fishers aren't informed of stock for their effort is in (1.28) to compare to cases with information about stock for fishers from managers to entice more participation in both fishery and invasive species goals.

$$p_L(qS_CE) = C_C(S_C)E. \tag{1.28}$$

Solving for a steady state of pure open access, $S_C = \frac{C_C(S_C)}{P_Lq}$. The shadow price for stock in Zone C has the standard form under open access (Conrad, 1999) as $\lambda_C = P_LqK_C + \frac{C_C(S_C)}{2P_Lq}$ to include in equations (1.24–1.27) of Zone B. Our empirical analysis compares management with and without updating information on the stock spatial transfer and damages with measures of stock over time and space in equations (1.27) and (1.28). This comparison helps move beyond a static solution of zero profits under open access, as it accounts for a mix of fishers in open access with information added on interdependence between zones through equations (1.16–1.23).

Information on damages and spatial transfers can alter fishers' incentives under open access, thereby engaging them in both invasive species management and fisheries activities. In the empirical analysis we explore how those components matter separately to finetune policy. A subsidy scheme in use from 2010 to 2018 by the Directorate of Fisheries aimed at fostering harvest of undersized crab in Zone C by commercial fishers. However, the subsidization policies do not explicitly internalize either damages or actual rate of stock transfer from Zone B. Our empirical analysis for open-access harvesting in Zone C will include such components for policy.

For non-cooperation we also examine how price, spatial transfer of stocks across zones and the nonmarket value of benthic habitat influence maximizing net benefits for fisheries revenues and Barents Sea habitats.

Empirical application

The following data are from historical cooperation through JointFish with an experimental fishery and the 2002–2006 restricted commercial fishery period when an agreement between the two countries was in place. Specifically, Table 1 depicts each country in terms

Table 1. Annual RKC catch 2002–2006, conducted under Russian-Norwegian cooperation. Unit: thousands of crabs

	Annual catch		TAC		Stock	
Year	Russia	Norway	Russia	Norway	Russia	Norway
2002	300	100	300	100	15,512	3,179
2003	600	200	600	200	19,995	3,575
2004	320	280	500	280	16,500	4,063
2005	1310	280	1,400	280	13,360	3,426
2006	1028	300	3,000	300	12,120	4,322

Source: Records from IMR (2017) and PINRO (2015).

of annual catch, and TAC including legal size males for commercial harvest in number of crabs (common unit during that period), drawing from IMR (2017) and PINRO (2015) measurements. We characterize binational coordination during that time with the stock amount set by both countries without economic optimization.

The empirical analysis has data sources described here with the model parameters listed in Table 2. The spatial transfer coefficients between zones are as follows: Pinchukov (2009) indicates 0.407 for the unidirectional westward transfer coefficient μ_A from source Zone A to receiving Zone B. The rate of RKC movement is 127.75 miles per year or 0.36 miles per day (Tal'berg, 2005). Windsland et al (2014) recorded data yielding an average of 0.002 for the transfer coefficient μ_B from source Zone B to receiving Zone C for the non-cooperative game based on field data collected during that timeframe. These coefficients are not density dependent (Windsland et al 2014). We consider the carrying capacity K_B for Norway's commercial fishery in Zone B, to be 55.21 mil. kg and for Zone C, carrying capacity K_C to be 10.39 mil. kg (IMR, 2017). The carrying capacity for Russia, K_A, is considered to be 389.15 mil kg (PINRO, 2015). The intrinsic rate of growth r is estimated at 0.227 (Falk-Petersen and Armstrong, 2013).

Norway's stock-dependent harvest cost function for the coastal fishery (Kourantidou, 2018) draws from three vessel classes' data all under 20 meters with pot gear, captured in logbooks. The estimation was based on data set of 52,325 fishing observations for the commercial fleet between 2002 and 2007. Vessel names and registry numbers were matched with records from 2016 available from the Norwegian Directorate of Fisheries. A stock-dependent Catch Per Unit Effort (CPUE) function was first estimated, capturing abundance, seasonality and vessel characteristics. The vessel length was evaluated constant at its mean using the classification of the annual profitability survey of the Norwegian fishing fleet, specifically vessels below 11m, and those between 11 and 14.9 m long (70% of the vessels were below 11m). Having established the relationship between CPUE and stock, accounting for individual vessel length, the data was aggregated to an annual fleet-level by creating a cost function weighted by vessel lengths each year. Published figures from the Norwegian Directorate of Fisheries for the average Norwegian fleet were used, where vessels are classified according to their length. These reported annual operational costs included both variable costs and fixed costs (i.e., fuel and lubrication oil, special taxes, insurance, maintenance, labor costs and depreciation). The fraction of crab quotas held by vessels, compared to the overall quotas of other species they held, was calculated using data from the 2016 catalogues of the Norwegian Fishery Directorate and adjusted for inflation. This resulted in a cost function in the range of 32 – 178/(S)$^{0.44}$ (in NOK). For the analysis,

Table 2. Parameters for analysis

Variables and Parameters	Value	Sources for Parameterization
p_H	372.78 NOK/kg	Norges Råfisklag (2007–2019)
p_L	251.22 NOK/kg	Norges Råfisklag (2007–2019)
K_A	389.15 mil. kg	PINRO (2015)
K_B	55.21 mil. kg	IMR (2017)
K_C	10.39 mil. kg	IMR (2017)
μ_A	.407	Pinchukov (2009)
μ_B	.002	Windsland et al. (2014)
r	.227	Falk-Petersen Armstrong (2013)
D_{FB}	23.49 NOK/kg	Kourantidou (2018)
D_{FC}	10.17 NOK/kg	Kourantidou (2018)
$\phi\ v_B(X_B)$	15.10–166.13 NOK/kg	La Riviere et al. (2014)
$\phi\ v_C(X_C)$	30.65-337.22 NOK/kg	La Riviere et al. (2014)
C_A	4.60 NOK/kg	Seung et al. (2015) and Abbott et al. (2010)
C_B, C_C	110.2 NOK 1000/vessel/ year	Kourantidou (2018)
q_A	1	Stesko and Bakanev (2021), Marine Stewardship Council (2020)
q_B	.75	Kourantidou (2018)
q_C	.75	Kuzmin and Sundet (2000)
M_B	10.78 NOK/kg	Norges Råfisklag (2019)
δ	Interest rate for discounting	World Bank and Norges Bank (2018)

an average of the cost functions from 2004 and 2005 was used, representing a lower bound compared to other years of the data set, $C(S) = 35\ NOK/(S)^{0.44}$. This choice was made based on the understanding that the crab fishery has inherently low costs, with expectations that experience and knowledge gained over the years would have further reduced these costs in the future. The processing cost, M, for live crab is 10.78 NOK/kg for several years of our analysis (2016–2019) (Norges Råfisklag, 2019). Granted, this cost may evolve due to value chain needs (Voldnes et al. 2020). Without data of recreational fishers' costs, we opt for using the average cost of the commercial harvest cost function from Kourantidou (2018) noted above for open access in Zone C with a cost offset from a government subsidy of 12 NOK/kg offered in this zone as an incentive to harvest for assumed undermarket sized crab. The catchability coefficient used for Zone B and C, (q_B *and* q_C), is 0.75 from research trawl surveys (Kuzmin and Sundet, 2000).

We do not have Russian cost data for a harvest cost function for Russia. Therefore, we use the costs of vessels from Alaska that joined the Russian fleet due to Alaska's crab rationalization program in 2005. The Russian fleet is more than 31 years old (Urner Barry

2020) and verifies that the same fleet from the 2005 transfer is still in place for generating a cost function. Seung et al. (2015) calculated an average expense per crab for fuel, bait and other crew provisions for each year 2006–2010 for the fleet with trap gear. Abbott et al. (2010) provide cost categories (labor, onboard), vessel capacity, catch and income per crew member for the harvest costs with annual vessel expenditures of labor, fuel, bait, provisions and quota lease costs to gauge CPUE by vessel characteristics and catch. We cite the preceding studies for vessels transferred from Alaska along with data on the vessels' volume of catch and revenues, boat size and crew size, and derive the arithmetic mean (fleet total over number of vessels by boat size and crew size) to quantify days at sea costs and per unit effort costs per volume of catch. Abbott et al (2022) modifies the earlier data and analysis of Abbott et al. (2010) with proprietary data from less vessels affecting Alaska harvest costs and contribution margins that have fallen following fleet rationalization. This accentuates the difference in costs between the Alaskan and Barents Sea fleets. The Russian harvest cost parameter, C_A, is estimated at 4.60 NOK/kg. The catchability coefficient for Russia, q_A, is assumed to be 1 (Stesko and Bakanev, 2021; Marine Stewardship Council, 2020). Stesko and Bakanev (2021) discuss research and fleet vessels for the coefficient and the Marine Stewardship Council (2020) refers to fleet vessels of Antey Sever LLC in the Barents Sea.

Prices for live and frozen crab are referenced from Norges Råfisklag (2019) and account for all types of crabs landed (male, female, different weight classes). Specifically, we take average prices in NOK per kilogram for live p_H (372.78 NOK/kg) and frozen p_L (251.22 NOK/kg) crab separately from period 2007–2019.

The empirical damage function includes market values of impacts on commercial species (bycatch and predation by RKC) along with nonmarket values for benthic impacts. Falk-Petersen and Armstrong (2013) included gear replacement and fuel costs associated with the bycatch impact. We reference more recent estimates for Norway's RKC fleet from Kourantidou (2018), with totals of 7,404,537 NOK for (additional) fuel (to reach crab free areas) and 1,858,165 NOK for maintenance and gear repair in 2015. Noting the fleet distribution and spatial variation in RKC across Zones *B* and *C*, the bycatch portion of D_{FB} is 6.85 NOK/kg for Zone *B* (2/3 of the total for fuel, maintenance, and gear repair) and for D_{FC} is 3.53 NOK/kg for Zone *C* (1/3 of the fuel, maintenance, and gear repair). The average interest rate δ was 2% for 2007–2019 (Norges Bank, 2018; World Bank, 2018).

The market value of RKC predation damages on commercial fisheries of lumpsucker and capelin draws from Kourantidou (2018), measured in value by kilogram. Losses from predation on commercial species, such as lumpsucker and capelin (Mikkelsen and Pedersen, 2012, are assessed by estimating the crab's predation impact through lower and upper bound estimates based on the proportion of landings potentially affected. Benthos degradation is assessed by evaluating the damage to benthic habitats resulting from RKC predation, incorporating estimates from nonmarket valuation studies (Groeneveld, 2010; Aanesen et al., 2015) and adjusting for the affected area, population, and foraging rates of annual benthic production in the region (Jørgensen and Spiridonov, 2013). These components collectively form the overall damage losses from the crab invasion, capturing both direct and indirect impacts. Given the differences in Zones B and C, we apply the upper bound estimates from Kourantidou (2018) for capelin and lumpsucker to Zone B and the sum of the lower bound estimates from Kourantidou (2018) for capelin and lumpsucker to Zone C. Numerically, the landing value sum of 6,572,524 NOK (capelin) plus 4,281,000 NOK (lumpsucker) from Kourantidou (2018) is 10,853,524 NOK, applied to Zone *B* estimate of landings and the lower bound of 88,420 NOK (capelin) plus 1,412,730 NOK (lumpsucker) is 1,501,150 NOK, applied to Zone *C* estimate of landings with less RKC invasion than Zone *B*. Predation value of 16.64 NOK/kg is added to

6.85 NOK/kg from last paragraph for D_{FB} and predation value of 6.64 NOK/kg is added to 3.53 NOK/kg from last paragraph for $D_{FC} = 10.17$.

The 2008 Marine Resources Act (Regjeringen Norges, 2008) includes Legally Protected Benthic Areas for seabed, corals, etc., described by Olsen et al. (2007). The Norwegian Ministry of Fisheries has introduced spatial closures to protect reefs, lobsters and benthos (ocean floor fauna and flora) from impacts of fishing [(Johnsen, 2017; Freiwald et al. 2004)]. These Norwegian policies enter into a nonmarket valuation survey (La Riviere et al. 2014) of the Norwegian public for benthic habitat values over space in zones around the RKC invasion (Zones B and C) for protecting benthic habitats. Therefore, the quantified nonmarket values of benthic habitat space are used instead of specific damages, such as to the Iceland scallop (Jørgenson and Primicerio, 2007) and soft bottom benthic habitats (Oug et al, 2011) that have not been valued to directly reference. The nonmarket public value of benthic habitat (equation (1.2)), for Norway are derived from the value Norwegians revealed through a discrete choice experiment survey (involving 4683 respondents). The choices of increasing beyond the existing 2,445 square km of protected benthic habitat area to either 5,000 or 10,000 square km resulted in willingness to pay values of \$85 per household for increasing to 5,000 square km and \$129 per household for increasing to 10,000 square km (La Riviere et al. 2014).

Fitting a curve from the La Riviere et al. (2014) results in a function relating area to monetary value and enables empirical estimation of the expected loss in value due to the change in the amount of benthic habitat area (the second component of equation (1.3)) for application to Zones B and C with spatial spread linking the zones. The spatial function is $7.346x + 521.89$ where $x =$ area in square kilometers. Note the following spatial area for each zone: Zone B has 20,054 square km and Zone C has 12,253 square km. For an average number of Norwegian households (2,272,730) during 2008–2017, the nonmarket value average is 1101 million NOK in Zone B and 1057 million NOK in Zone C. Jørgensen and Spiridonov (2013) estimated biological changes in annual benthic fauna production due to RKC foraging with several ranges in predation rates over space. For ς_i, the variance component of equation (1.3) we reference the measure of crab impact from Jørgensen and Spridonov (2013) taking the average (22.5%) from their widest range measure of 5%-40% and link it to the spatial benthic values of the Norwegian public for both zones. This procedure helps account for the variation drift term in the damages value (equation (1.3)). The range for Zone B is 23 million NOK with minimum 2% predation and 243 million NOK with 22.5% predation. Zone C's range is 21 million NOK with minimum 2% predation and 231 million NOK with 22.5% predation. The range of values for benthos in each area per kilogram of crab based on an average volume for each area over the time period of non-cooperation of 15.10 NOK to 166.13 NOK per kilogram of crab for $\phi v_B(X_B)$ in Zone B and 30.65 NOK to 337.22 NOK per kilogram of crab for $\phi v_C(X_C)$ in Zone C.

The magnitude of commercial and nonmarket components of damage combined is less than the commercial price for RKC per kilogram. Damages approach half of the value per kilogram of commercial price for live crab in Zone B and three quarters of the value per kilogram of the commercial price for live crab in Zone C among values in our applied optimization.

Figure 3 depicts Norwegian landings under the MSY policy in Zone B (E. Finnmark) and open access in Zone C (W. Finnmark) with data from Norges Råfisklag (2019) and IMR (2018). Norges Råfisklag data for price per kg in both zones follows the same upward trend per kg, accounting for high value males in the individual vessel quota Zone B versus more varied crab landed in the open-access Zone C.

Russia utilizes two locations to supply 70% of the global RKC market. The Russian portion of the Barents Sea is one location contributing 40% (42.6 mil. Kg) of Russia's total

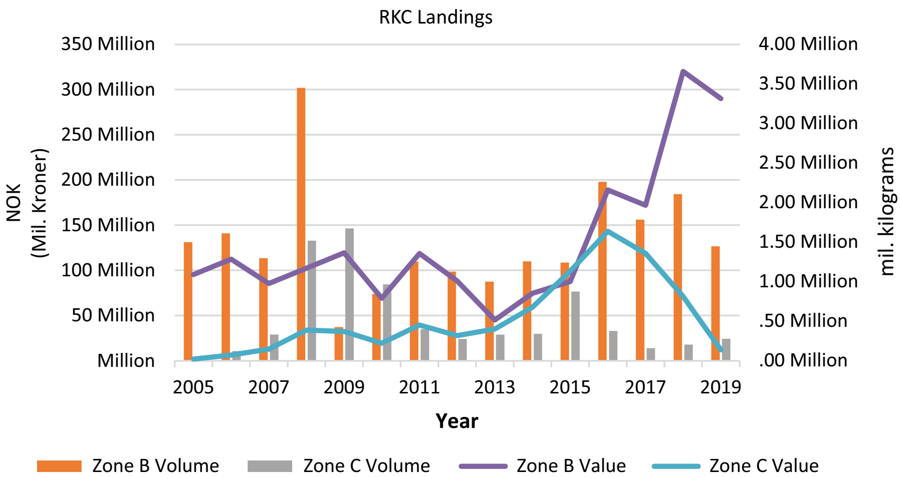

Figure 3. RKC landings in Norway.

supply in years 2016-Spring 2020 (Urner Barry, 2020) The Russian Far East, adjacent to the Bering Sea is the second location contributing 60% (64.4 mil. Kg) of Russia's total supply. Proximity to Asian markets and the higher price for live RKC have contributed to the steady rise of the Far East Russian supply. From the preceding discussion of empirical data, we calibrate the analytical model to derive optimization results for both countries managing the RKC independently. Note, Norway's Barents Sea supply is live RKC, with landings of 1,135 mil. kg live and 653 thousand kg frozen for 2019, an increase of 3% from 2018 (Urner Barry, 2020).

Results

During 2002–2006 both countries agree on harvest restrictions without enabling coordinated, optimized harvest as a function of the stock and movement across space and time. The percentage changes in stock (directional bars) in Figure 4 show large interannual variation, switching from positive to negative stock changes each year for both countries between 2002 and 2006 levels shown in Table 1 without moving towards a stabilized stock.

Results for each country under non-cooperation (after 2006) span scenarios with and without measures of stock spread across zones and values for benthic habitat. Through these scenarios we explore the role of information updating feedback Nash equilibrium strategies.

For Zone *A*, an increasing trend occurs in Russia's RKC harvests in the Barents Sea from 2010–2012 of 4000–5000 metric tons, 5500–6400 metric tons in 2013–2015 and in 2017 and 2018 are 8000 metric tons (verified by Dvoretsky and Dvoretsky, 2018 and 2022) under the full information scenario of including spatial transfer μ_A, from Pinchukov (2009). We apply the non-cooperation constrained optimization from equation (1.6) where supply chain costs are higher than in Russia's Far East and the Barents Sea frozen RKC price. Results for Russia's Zone *A* depicted in Figure 5 show stock changes are larger

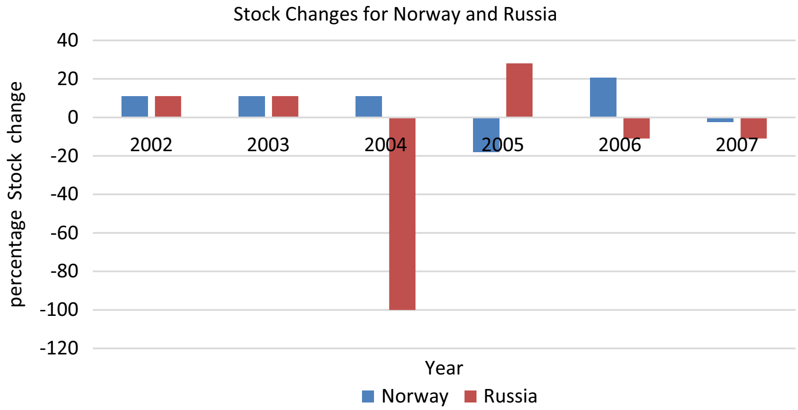

Figure 4. RKC stock changes during 2002–2007 in Norway and Russia.

initially (95% stock change decline in 2008 prior to 1% stock change decline for rest of time horizon) when including component μ_A compared to the scenario without including stock spatial transfer information between Zone *A* and Zone *B*.

Figure 5 has colored bars depicting percentage stock change, noting positive and negative directions of percentage stock changes relative to the previous year in the time horizon. In general, the stock changes reduce from the large range swings occurring during 2002–2006, prior to separate management by Russia and Norway. Optimizing RKC management in absence of information on spatial transfer μ_A yields an initial 16% stock change decline in 2008 and switches to a continual percentage stock change increase. These stock change results confirm Zone *A* is a backstop RKC reserve at the lower frozen price compared to a separate RKC Far East stock that is closer to the Asian market for the higher price live RKC.

Price differences between the Far East and Barents Zone *A* stay the same even after a certification of Russian Barents Sea RKC by the Marine Stewardship Council (MSC) happens in February 2018, with frozen RKC price lower than Far East live RKC price. Urner Barry does not distinguish between Russian MSC certified RKC and non-MSC certified RKC in their market reports (Schrieber, 2020) thereby perpetuating the Far East as the primary supply for Russian RKC at the higher price. Empirical conditions and our results imply that Russia has price and spatial transfer as incentives to increase its harvest in Zone *A* and reduce stock abundance.

Our optimization results for Zone *A* are validated by the biological study of Dvoretsky and Dvoretsky (2018) for the commercial stock, catch per ship per day and catch per trap both decline after 2007 (harvests are below quota). We can compare our optimization results to two biological stock assessment methods from Russia. Russia uses both the Collie-Sissenwine Method, which extracts a stock abundance signal from noisy catch survey analysis, and a second method, the Leslie Depletion Model for fisheries stock assessment in condensed space compared to the Collie-Sissenwine Method (Acoura Marine, 2017). Both methods align with our results that indicate a positive rate of change in stock over time. However, the Collie-Sissenwine Method is consistently 50% higher in magnitude than the Leslie Depletion Model and our results range in between both methods for stock transfer connectivity included to account for spatial area beyond condensed space (Leslie Depletion Model) in a unidirectional manner aligned with the stock dynamics.

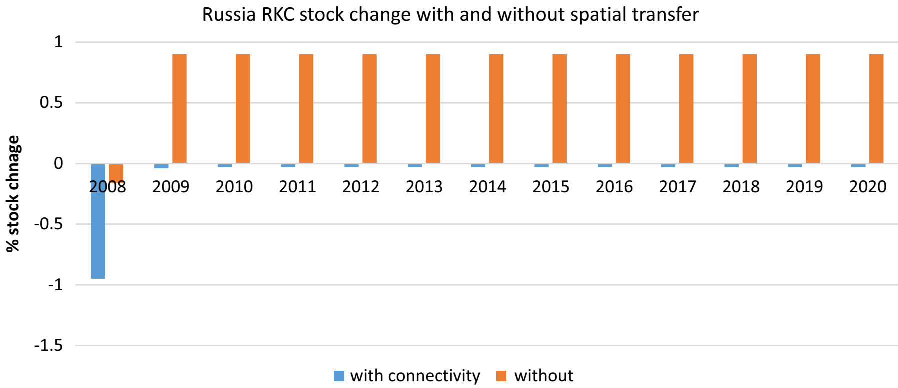

Figure 5. Results for Russian RKC optimal annual stock changes under varying assumptions for spatial transfer.

Without including stock transfer between Zones *A* and *B*, our results are closer to the Collie-Sissenwine Method.

The stock equation (1.1) shows continued RKC stock growth with restricted harvest in Zone *A* facing a lower market price than the live RKC price in the Far East. The Russian Barents Sea RKC stock persists dynamically and spatially under conditions parallel to both demand and supply changes in the international market post Spring 2020 due to COVID-19.

Figure 6 shows results for Norway's Zone *B* quota fishery. Optimization scenarios that integrate information on stock spatial transfer and damage values lead to smaller stock fluctuations over time compared to before 2007–2008. Other analyses in fishery economics literature assert smaller stock fluctuation helps to sustain a fishery (Costello and Kaffine, 2008). Norway has formally stated a goal of a stable long-term fishery for Zone *B* (Sundet and Hoel, 2016); Figure 6 shows at least one dynamic path of stock change from optimizing with full information achieving that goal among the scenarios depicted with various colored bars. Norway implemented a higher TAC immediately after 2007 with the change to non-cooperation, as shown in the landings data in Figure 3. Figure 6 includes results for Zone *B* stock change post 2007 under non-cooperation with and without information on Zone *C*'s shadow value, spatial transfer with Zone *A* and *C*, and damage values included in Zone *B* harvest quota decisions. Purple bars in Figure 6 depict the percentage changes in stock tied to the total allowable catch set by managers from biological stock assessment only, which is the current management. The magnitudes and signs of purple bar stock changes are parallel to before 2008 without information included on spatial transfer nor ecological values. There are large interannual variations, switching from positive to negative percentage stock changes. The orange bars in Figure 6 depict percentage stock change in Zone *B* including quantification of μ_A by Pinchukov (2009) of spatial transfer between Zone *A* and *B* along with damages for Zone *B*. The orange bars exhibit negative percentage stock change for the live crab fishery reducing stock when non-cooperation removes restrictions prior to 2007 that leads to harvest upswing.

Non-cooperation yields a consistent 12% stock change decline in RKC with the full information about stock spatial transfer measures between Zones *A* and *B* as well as between Zones *B* and *C* and damages in Zones *B* and *C* depicted by blue bars in Figure 6, typically 1% more stock change than the scenario without considering zone C

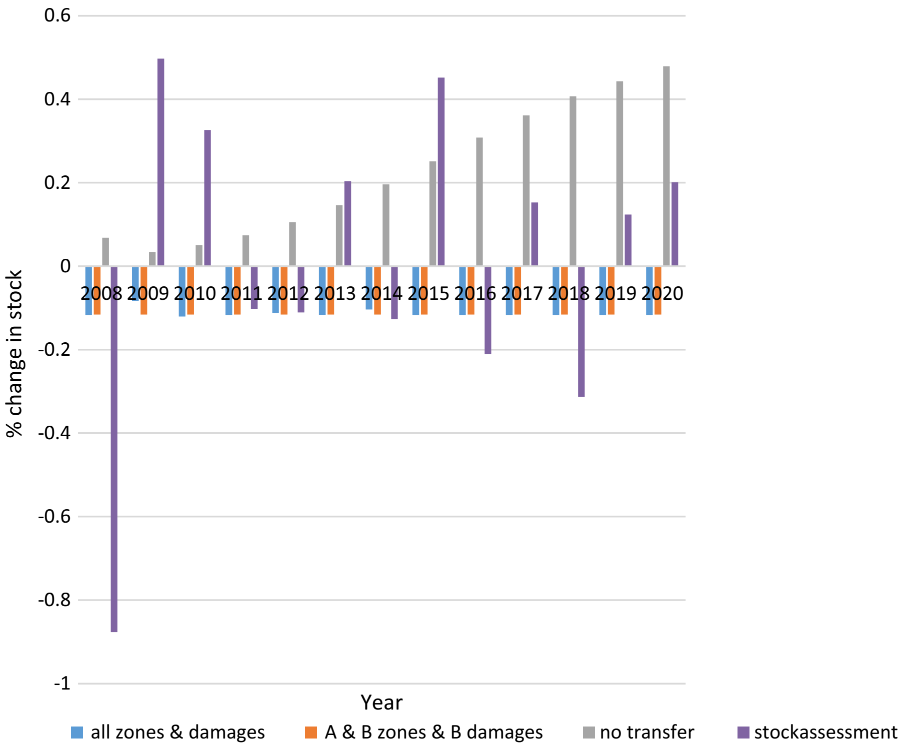

Figure 6. Results for Norwegian RKC optimal annual stock changes in three scenarios varying spatial transfer and damage values.

(orange bars). This scenario of all those we consider, internalizes spatial transfer measures between zones known after the start of non-cooperative management as well as damages for management in both Zones *B* and *C*. Therefore, optimal management balances fishery and habitat goals driving the strong incentives to harvest, including the shadow value of Zone *C*. This shadow value embodies the market value of Zone *C* RKC along with spatial transfer and with market and nonmarket value damages, noted in equation (1.26) leading to a reduction in stock variation, theorized by Costello and Kaffine (2008).

Gray bars in Figure 6 depict the scenario including information on both zones' damages without information on spatial transfer through dispersal nor shadow value of Zone *C*. These conditions result in steadily increasing stock change. The price for live RKC affects optimal management and harvest decisions. It reflects a steady international market that has rebounded through 2020 after an initial major decline from COVID-19 (Kjolberg, 2021).

Within Zone *C*, we compare open access with and without information from Windsland et al (2014) of stock spatial transfer from Zone *B* and the value of RKC damages (market, subsidy and nonmarket values). Typically, open access has zero rents and search friction causing a congestion externality disincentive (Gordon, 1954). Baseline open-access harvest is 1.45 mil. kg with the stock below carrying capacity at 10.19 mil. kg but above MSY and this is the without information case. Our sensitivity analysis starts with one variation from the open-access baseline including value of damages as an additive term to the open-access solution from equation (1.28), resulting in increasing RKC harvest by 46%. The increase approaches the RKC landing levels from early stages of non-cooperation. Including damage values reduces the optimal stock by 31%. Policy context for this scenario is plausible with the documented landings in Figure 3. The Norwegian Marine Resources Act provides context for to include the nonmarket valuation generated from surveying the Norwegian general public about protection zones for habitat value.

The second variation from the open-access baseline includes information from Windsland et al (2014) measure of stock spatial transfer. With spatial transfer to Zone *B*, the harvest is 52% higher than baseline from the marginal stock effect as a change from baseline open access of average costs and benefits. This scenario reduces the stock by 39%, a larger change than from the addition of market and nonmarket damage values only. In both of the added information scenarios, incentives to increase harvest result in stock decline. Specifically, harvesters are motivated to avoid typically zero gains and congestion under open access, through the stock transfer. The decline in the stock helps to address Norway's commitments to the Convention on Biological Diversity. RKC management and policy would benefit from information about spatial transfer from Zone *B* to improve outcomes as shown to enable both fishery and ecological goals.

Our results indicate information on damages and stock spatial transfer helps along with price and subsidy as plausible incentives to engage public participation in Zone *C*'s open-access management. Full transparency through public disclosure of these components may ensure public participation in open-access strategies for managing RKC, thus increasing harvest levels and slowing the invasion at the frontier. Given our sensitivity analysis ties the ecological damages to the nonmarket value survey of the Norwegian public and the policy context of the 2008 Marine Resources Act, transparency of the damages appears necessary along with available price information to prompt RKC stock change through public action.

The Directorate of Fisheries has used subsidies to commercial fishers since 2010 to encourage harvest in the open-access area. Our results imply such subsidies may be small compared to taking market and nonmarket damages and stock spatial transfer into account that increase the benefits of harvesting more compared to the open-access level of RKC in Zone *C* without accounting for these incentives to entice participation to help accomplish the ecological goal.

Both Zones *B* and *C* operate in tandem towards Norway's two goals that maximize different benefits: maintaining a fishery and protecting benthic habitat. Our simulation shows Norway's zones are managed with market price-driven harvest incentives when fully informed by stock spatial transfer to help overcome the congestion disincentive (zero profit) from the open access in Zone *C*. Our analysis shows the optimal gain from jointly accomplishing objectives of fishery's profitability as well as mitigation of the invasive species. We do this by exploring non-cooperative Nash strategies that account for stock spatial transfers, shadow values along with market and nonmarket damage values for both Zones *B* and *C* to achieve fisheries and habitat benefits through joint stock change management instead of optimizing one goal and one zone. Dynamic non-cooperation is more responsive with self-enforcing incentives than previous restrictions (before 2007) if updated information about stock spatial transfer is included. Such information helps management capitalize on a wider sovereign space (Zone *C*) while possibly weathering abrupt disruptions in the RKC supply chain and market demand from unexpected phenomena such as COVID-19. The pandemic halted stock assessments that offer data from surveys that are conducted to understand species populations in terms of size and geographic extents. Our analysis recognizes the benefit of simultaneous monitoring of RKC and benthos in the annual survey to help integrated analysis and adaptive management.

Discussion and conclusions

Our applied analysis integrates asymmetric economic components (prices, costs) and ecological components (ecosystem market and nonmarket values, stock transfer) in a transboundary setting experiencing trade-offs in fisheries and invasive species management over time and space for Barents Sea RKC. Empirical results from our dynamic bioeconomic model compare non-cooperative Nash strategies for potentially balancing RKC fishery management and habitat goals with market and ecological incentives. We explore the role of information in updating dynamic and spatial management strategies we simulate integrating economic incentives, nonmarket values, and stock spatial transfer across three different zones. We find the ability of reducing stock variation through multiple incentives including prices. High prices in RKC markets are a prerequisite for market-driven harvest to help net benefits of fisheries and habitat conservation goals. Harvesting RKC serves as mitigation and adaptation for invasive RKC in the Barents Sea both in designated fishery and open-access zones when measures of spatial transfer are nested in management. Results show both Russia and Norway sustain RKC stocks for differentiated commercial markets. Inclusion of Norwegian public values for benthic habitat protection in our analysis aligns with the 2008 Marine Resources Act precedent for precaution. Our analysis suggests integrating full information of quantified measures across space and time improves varied management for fisheries and habitat protection goals in Norway's zones. Our bioeconomic model helps to understand trade-offs that can inform management to balance fisheries and invasive species control by including components of damages and spatial stock connectivity along with price incentives in all zones. These components address the three pillars that Grafton and Kompass (2014) have suggested for a general fisheries and ecosystem balance. These pillars are significantly more narrow than the three pillars of sustainability (environment, economics, society) which has been investigated by Asche et al. (2018), who show that the strength correlation between the pillars vary with management system. Clearly, the zones vary as our analysis shows the incentives for each sovereign country competing in the international RKC market can help self-enforcing management in absence of negotiation between them, noted in theory by Dockner and Long (1993).

Since management involves participation of more than commercial fishers in Zone *C* for deliberate RKC harvest and stock change, enabling awareness of the fishers through providing available information to invoke optimal incentives for participants (commercial and recreational fishers) in the zones facing quota and open-access management. Besides any initiatives to help reduce harvesting costs in Zone *C* (gear exchange, survey of fishing costs), formally including information on dynamic stock transfer across zones may catalyze a beneficial incentive for more RKC harvest and fishing participation than typical open-access congestion prompts in order to limit the spread of RKC. Depicting the stock change to compare different scenarios is a fundamental gauge for both fishing commercial value and RKC presence in zones tied to ecological habitat values.

Enabling public information about RKC stock spillover into Zone *C* would be a way to match the scale of policy to the scale of RKC spread following suggestions in the property rights literature (Frischmann et al, 2019). Other dissemination tools that can be used to increase participation are the annual Barents Spectacle where IMR scientists present their studies to the general public, as well as real time information on fishing pressures and crab sightings gleaned from BarentsWatch.no, where fishers note their activities to avoid entanglements.

Ending incomplete coordination in 2006 between Norway and Russia for RKC in the Barents Sea has resulted in more consistent stock change for each country to manage than the large interannual stock change variation from before, as our analysis demonstrates. Perfect competition in the RKC world market prompts the RKC fisheries in both countries to be managed for the long run with strong incentives from current RKC price increases (for both frozen and live). Norway may employ *in situ* inventory (aligned with forecasted RKC spread) to smooth timing and volume from supply and demand disruptions while just in time supply remains part of Norway's strategy to compete in live RKC. Evidence shows both countries aim to move on from the temporary halt in RKC trade in 2020 with increases in their 2021, 2022 and 2023 quotas. COVID-19 halted Norwegian maritime activities at ports and in fisheries during the first quarter of 2020 with a 40% drop in price (from $42/kg to $25/kg), and a resulting direct drop in harvest that matches the decline of an 18% reduction in volume (422 metric tons exported) compared to 2019 (Hodges, 2020). The RKC stock over time and space keeps evolving with a temporary delay in harvest coupled with capacity issues revealed by COVID-19 (Hodges, 2020). Disruptions to the supply chain, such as those stemming from the COVID-19 pandemic or Western sanctions against Russian crab cannot be countered with a quick shift from live to frozen RKC because these disruptions would require ample capacity for onshore inventory. *In situ* cold storage at sea remains the feasible and cost-effective technology for both live and frozen RKC supplies of Russia and Norway. Such storage keeps stock higher, exacerbating the associated negative externalities.

Our analysis shows incentives for balancing both habitat value and fisheries under non-cooperation depends on economic incentives as well as information pertaining to spatial stock dynamics and benthic habitat values. The forecasted spread of RKC in sovereign Norwegian waters noted in our introduction supports Norway's fisheries goal in the long run. The zones exhibit management over time and space with potential to adapt through integrating components we analyzed, that may help in responding to shocks not forecasted, such as COVID and Western sanctions on Russia.

We sought to include available measures (stock, ecological values, etc) and recognize the need for ongoing effort to keep updating such information and support all efforts for contributing to adaptive management. Ultimately, that will help the ongoing challenge to balance long run fishing and ecological goals.

Data availability statement. We have utilized publicly available data from sources cited in the manuscript and can make that data available upon request.

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