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Evaluating the effectiveness of real-time closures for reducing susceptibility of small fish to capture

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Real-time spatial management in fisheries, a type of dynamic ocean management, uses nearly real-time data collection and dissemination to reduce susceptibility of certain species or age classes to being caught in mixed fisheries. However, as with many fisheries regulations, it is difficult to assess whether such a regulation can produce tangible results on population dynamics. In this study, we take advantage of a rare opportunity in which data regarding real-time closures (RTCs) are available for 1990–2014 alongside annual estimates of fishing mortality for three species (Atlantic cod, haddock, and herring) and catch for four species (all plus saithe) in Icelandic fisheries management. We use time series analyses to assess whether RTCs work as expected and yield a lower susceptibility of small fish to being caught, indicated by lower catch levels and selectivities (as estimated from fishing mortalities) in years with more closures. Results indicate that haddock and herring followed this pattern, but only under conditions of generally high fishing mortality. This study represents the first time evidence has been presented that real-time fishery closures can have a beneficial effect on population dynamics, but also suggests that results differ among species.

Keywords: dynamic ocean management, fisheries management, real-time area closures, selectivity, spatial management.

Introduction

Spatial management has long been used in fisheries to reduce discards and bycatch, thereby increasing fishery selectivity or protecting important habitat (Dunn *et al.*, 2011). When designed appropriately, fishery closures that are static in time and space can be effective management tools (O'Keefe *et al.*, 2014). However, they have also been criticized as ineffective, often due to the mismatch between the spatial and temporal scales on which closures are drawn vs. the scales on which resource users and ecological processes operate (Bailey *et al.*, 2010; O'Keefe *et al.*, 2014; Maxwell *et al.*, 2015; Dunn *et al.*, 2016). Dynamic ocean management (DOM) is a promising solution to this mismatch, as it is "management that uses near real-time data to guide the spatial distribution of commercial activities" (Lewison *et al.*, 2015, p. 486). It can theoretically respond faster to changes in the ecosystem than can typical static management schemes, which generally assimilate data on an annual basis (Dunn *et al.*, 2014, 2016; Kraak *et al.*, 2014). As technological innovation speeds and eases the burdens of data collection and sharing, DOM has become an increasingly viable option to facilitate adaptive ecosystem-based management (Lewison *et al.*, 2015; Maxwell *et al.*, 2015; Dunn *et al.*, 2016).

DOM includes mechanisms such as spatial real-time fishery closures (RTCs), move-on rules (Dunn *et al.*, 2014, 2016; Bjorkland *et al.*, 2015), risk pools (Holland, 2010; Holland and Jannot, 2012), and real-time incentive-based management (Kraak *et al.*, 2014, 2015). The actual implementation of such mechanisms vary widely, from top-down government-sponsored regulations to industry-sponsored voluntary programs, as well as co-management arrangements that fall in-between (Little *et al.*, 2015). RTCs have gained popularity in the past decade as a fisheries management tool for reducing discards or avoiding hotspots of unwanted species (O'Keefe *et al.*, 2014; Lewison *et al.*, 2015; Little *et al.*, 2015). Real-time oceanographic or catch data may be used to fit spatial models aimed at predicting the occurrence of species to avoid, or in other instances, relatively simple data are needed (Lewison *et al.*, 2015). For example, RTCs are often event-triggered, such that a single reported high catch of a by-catch species or age class will induce a no-fishing zone for several weeks. Similarly, move-on rules require that fishers move a certain distance before continuing operations (Dunn *et al.*, 2014, 2016; Bjorkland *et al.*, 2015). In top-down orchestrated closures, reported catch data are generally gathered from observer coverage or inspections; in voluntary situations, personal catch data are exchanged for access to larger pools of collective catch data and/or quota (Little *et al.*, 2015).

Although DOM has the benefit of reacting quicker to stock dynamics and is a clear example of adaptive management, it has not been widely implemented because it poses a variety of challenges for success (Hobday *et al.*, 2014; Lewison *et al.*, 2015). First, data must be collected, potentially processed, and finally disseminated on fast time-scales, which may be relatively labour-intensive and costly for both industry participants and managers (Lewison *et al.*, 2015; Little *et al.*, 2015). Second, it often relies on stakeholders and managers to work together to set up a system in which resource use data are shared, requiring negotiation and agreement on the terms of use (e.g. Holmes *et al.*, 2011; O'Keefe and DeCelles, 2013; Wallace *et al.*, 2015), within a context where laws and policies support such use (Hobday *et al.*, 2014). Third, the correct incentive structure must be in place in order for the benefits to appear great enough for both initiation and compliance (Holmes *et al.*, 2011; Hobday *et al.*, 2014; Lewison *et al.*, 2015). Finally, although it has a greater *potential* to match spatio-temporal ecosystem dynamics than static annual closures, some knowledge regarding ecosystem dynamics (e.g. site fidelity or movement patterns) is needed to optimally design the closure system for the purpose at hand (O'Keefe *et al.*, 2014; Kraak *et al.*, 2015; Dunn *et al.*, 2016).

Therefore, it is useful to know whether DOM is effective, and under what terms of implementation. Overcoming the first three challenges indicates that a program can be initiated and is at least operating as intended. Clearly, voluntary programs yield incentives for negotiation and participation (second and third challenges) or else they would not exist. For example, participants of risk pools use more of the high-value quotas without using more constraining species quota, which would risk an early fishery-wide closure (Labrum and Oberhoff, 2013; De Alessi *et al.*, 2014). However, evaluating the fourth challenge requires linking the programme function with intended ecological effects. For example, some programmes can be deemed ecologically effective through demonstrated avoidance of habitat or bycatch, while simultaneously beneficial to fishers by enabling retention of catch that would have been foregone, avoidance of nuisances in catches, or avoidance of penalties (Dunn *et al.*, 2014, 2016; Bjorkland *et al.*, 2015; Lewison *et al.*, 2015; O'Keefe *et al.*, 2014; Dunn *et al.*, 2016).

In the cases of long-standing government-sponsored regulations, there is often less impetus to evaluate whether a regulation is ecologically effective, as long as it is perceived to be beneficial with no negative side effects (i.e. "if it ain't broke don't fix it"). Evidence for effectiveness in this form may be simply that the system is at least operating as intended, without analysing ecological consequences. For example, pilot projects indicate that fishers do indeed move from areas of higher to lower concentrations of the bycatch species when given catch information (Needle and Catarino, 2011; Bethoney *et al.*, 2013). Evidence for effectiveness in reducing fishing mortality or other environmental impacts, on the other hand, is more difficult to trace due to the lack of a controlled experiment: typically no fishing occurs within closures to test the closure's effectiveness (Bailey *et al.*, 2010). Closures may be ineffective if fishers move straight back into closures after re-opening or the closure length is too short (Needle and Catarino, 2011). Furthermore, evaluating the effectiveness of any regulation in fisheries is often difficult because of confounding factors, such as simultaneously introduced regulations or economic and environmental changes. In Norway, a long-standing system of RTCs, along with effort reduction and favourable environmental conditions, are thought to have contributed to the recovery of cod in the Norwegian and Barents Seas, but these three factors cannot be disentangled (Johnsen and Eliasen, 2014; Eliasen, 2014). Nonetheless, there exists some limited evidence for reductions in catch of certain species or age classes (Bailey *et al.*, 2010; Holmes *et al.*, 2011; O'Keefe and DeCelles, 2013; O'Keefe *et al.*, 2014), and a theoretical study indicates that real-time management may be even more effective at controlling harvest rates than traditional annual harvest control rules (Kraak *et al.*, 2015).

In no cases, however, has an RTC system been linked to population dynamics, thereby yielding direct evidence for fulfilling its ultimate mandate. In this study, we attempt to do just this by testing whether there is a correlation between usage of the Icelandic RTC system and fishing selectivity or catch of young fish. RTCs have occurred in Icelandic fisheries as far back as the late 1960s for the Atlantic cod *Gadus morhua* and 1970s for the Atlantic herring *Clupea harengus*, although other species generally have closures beginning in the late 1980s. Today, the Icelandic RTC system applies to nine demersal groundfish species, four pelagic fish species, Norwegian lobster (*Nephrops norvegicus*), and shrimp (*Pandalus borealis*) (Schopka, 2007). In most cases, to trigger a closure, catch of a certain size range of the species exceeds a certain percentage of the total catch (e.g. 25% under 55 cm for cod). Closures are generally detected by onboard observers, who subsample and measure the catch in 15–20% of the cod fishery (Little *et al.*, 2015), although other mechanisms are possible. The Icelandic Marine Research Institute (MRI) then has the authority to define a closure area, which must be advertised on radio by the coastguard and the broadcast by the government radio station before the closure takes effect. The closure can take effect within less than an hour from advertisement, and lasts for 14 days, but may be extended for another 7 days if necessary.

To properly assess whether the RTC system works (i.e. whether it is having ecological effects), it would be best to test differences in mortality before and after implementation of a regulation, or between stocks or areas that differ in a specific regulation, while controlling for all other covariates. However, the introduction of a regulation as a controlled experiment is exceedingly rare in management systems in fisheries. As a result, the task often relies on correlations, as is the case in this study. Luckily, extensive time series of records regarding Icelandic RTCs allow for enough data to test for correlations. In most other locations, RTCs have only been implemented in the past decade; only RTCs in the Barents and Norwegian Sea cod and haddock fishery have been implemented for a comparable length of time (Little *et al.*, 2015). As a result, the Icelandic system yields an unprecedented opportunity to analyse a unique dataset. Here, we add to the literature surrounding RTCs by providing the first analysis of a direct effect of RTCs on catch and/or selectivity of four species in the

Icelandic fishery: Atlantic cod, haddock (*Melanogrammus aeglefinus*), herring, and saithe (*Pollachius virens*). These species were chosen because they all have a consistent record of relatively high numbers of closures within the last 30 years, as well as data-rich stock assessments that yield time series of fishing mortality and biomass estimates. We test for a relationship between the numbers of RTCs and catch levels (adjusted by biomass) or selectivity-at-age estimates (calculated from fishing mortalities) for the youngest age class possible of each species. A negative relationship would support the idea that RTCs reduce susceptibility of small fish to being caught. We also include two covariates, the yearly effect of fishing mortality (which we term "relative total effort") and adult biomass, in an attempt to control for possible confounding factors associated with changes in fishing pressure and the general state of the fishery.

Methods

Data sources

Databases at the MRI were used to obtain data on the number of RTCs ($R_{t,s,g}$) by calendar year (t), species (s), and gear (g). Trawls, longlines, and gillnets caught the vast majority of cod, haddock and saithe, and comprise roughly <1% of those data. Bottom trawls, mid-water trawls, and demersal seine were summed to form the "trawl" RTC category; only longline composed the "longline" RTC category. Handline RTCs were excluded. RTC data for herring were not split by gear as they generally included only herring-specific or mid-water gears (i.e. herring trawls, herring nets, herring or non-specific mid-water trawls, and purse seines). Not enough gillnet RTCs occurred for any species for analysis, due to very low selectivity for small fish.

Survey abundance indices at age (a) were taken from acoustic surveys for herring and spring groundfish surveys for all other species ($I_{t,a,s}$). Catches from all gears ($C_{t,a,s}$), weights ($W_{t,a,s}$), and fishing mortality estimates ($F_{t,a,s}$) were obtained from stock assessments completed by the MRI. These data were also available in national annual stock assessment and ICES reports (Anonymous, 2015; ICES, 2015). Survey abundance indices were transformed into reference biomass indices after multiplying by weight-at-age and summing over the appropriate age range of available survey data ($B_{t,4+,s}$ for cod, saithe, and herring, and $B_{t,3+,s}$ for haddock). Fishing mortality estimates were decomposed into an age-separable component of fishing mortality (here referred to as selectivity, $S_{t,a,s}$) and a yearly scaling component of fishing mortality (here referred to as relative total effort, $E_{t,s}$) within each species, by dividing by the selectivity within a year via the equation $E_{t,s} = F_{t,a,s}/S_{t,a,s}$. The selectivity of the fully selected age(s) within each year then equalled 1, and relative total effort for that year was the maximum fishing mortality value. Relative total effort therefore scales overall fishing mortalities across years, partially indicating differences in fishing effort as estimated in the stock assessment models, although it is not a true measure of fishing effort. Similarly, age-specific selectivity varies across years and reflects the degree to which a certain age class was susceptible to being caught relative to other age classes within that year, due to gear, fish vs. fishing distributions, or other factors.

Background on Icelandic fisheries

In Iceland, herring was the first species to have ITQs established in 1979 (Jakobsson and Stefánsson, 1999), although most species have been managed under an individual transferable quota (ITQ) system since 1991 (Arnason, 2005). Fishers are required to keep detailed logbooks for each haul/set. There appears to be little unreported catch as landings generally agree with logbook-based catches, although there are still low levels of discards in the form of high-grading for the largest stocks (0–5%, ICES, 2015; Pálsson *et al.*, 2015;). Annual stock assessments are completed in conjunction with the International Council for Exploration of the Seas (ICES), resulting in total allowable catches. There is a full discard ban, and there are technically no minimum landing sizes, but gear restrictions and the RTC system are intended to reduce selectivity on small-sized fish (ICES, 2015). The number of RTCs has greatly increased over the time period analysed (Figures 1–4, panels a). Permanent time/area closures are likewise utilized to protect spawning or juvenile aggregations, and a series of consecutive RTCs may allow the Minister of Fisheries and Agriculture to establish a longer closed area (Schopka, 2007). Further details can be found in Supplementary Material Section S1.1.

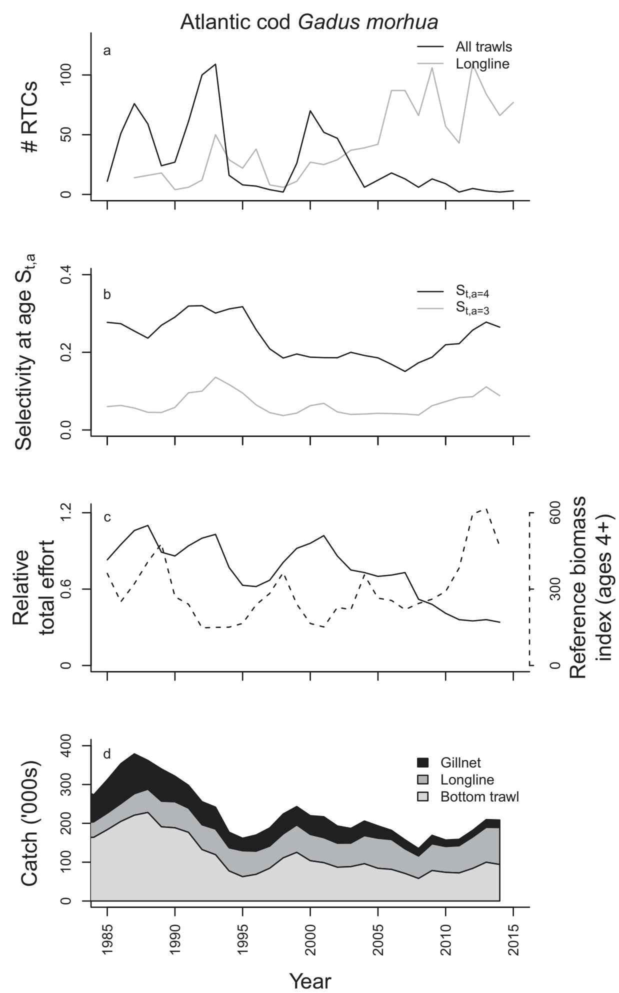

Figure 1. Time series of numbers of RTCs by gear (a), selectivity estimates for ages of recruitment (3) and entry (4) (b), relative total efforts (left axis) and reference biomass indices (right dotted axis) (c), and catch numbers by gear (d) are shown for cod.

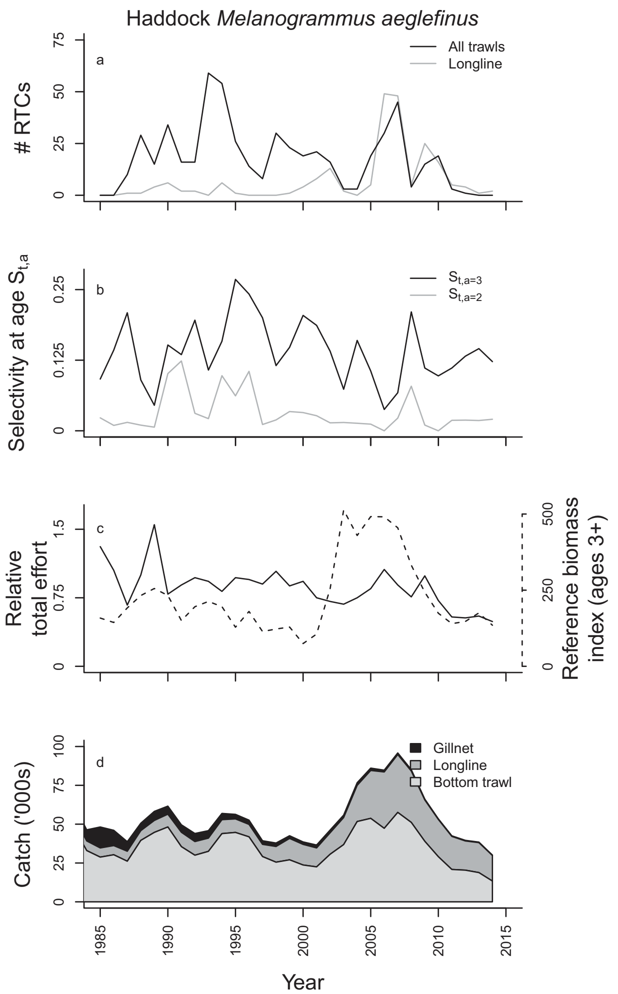

Figure 2. Time series of numbers of RTCs by gear (a), selectivity estimates for ages of recruitment (3) and entry (4) (b), relative total efforts (left axis) and reference biomass indices (right dotted axis) (c), and catch numbers by gear (d) are shown for haddock.

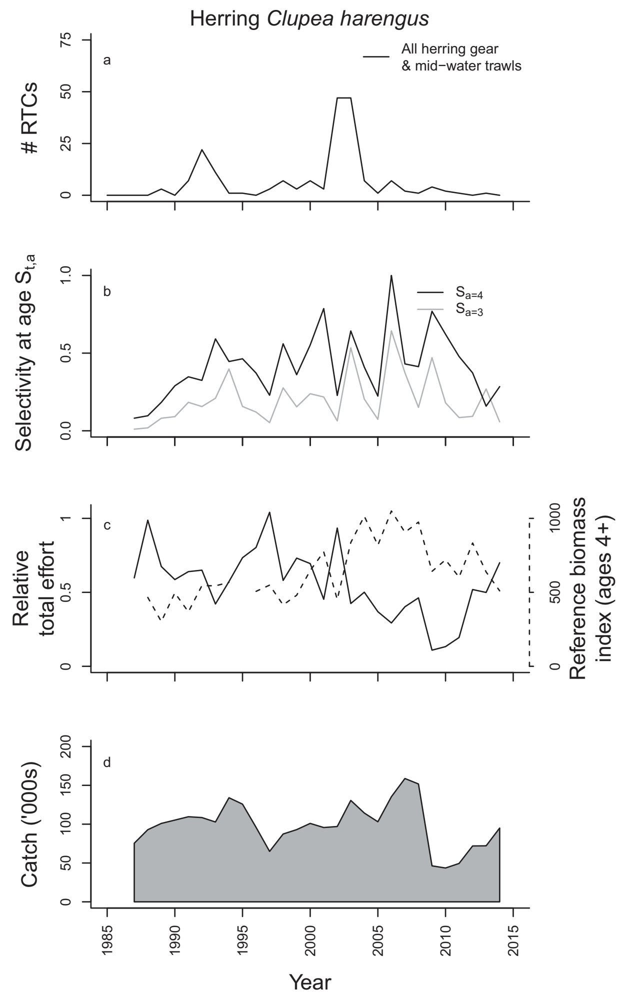

Figure 3. Time series of numbers of RTCs by gear (a), selectivity estimates for ages of recruitment (3) and entry (4) (b), relative total efforts (left axis) and reference biomass indices (right dotted axis) (c), and catch numbers by gear (d) are shown for herring. Survey data are missing from 1995.

Cod

Substantial effort reductions began in the 1990s (ICES, 2015) and a harvest control rule was introduced in the 1995–1996 fishing season. Subsequent modifications to the harvest control rule decreased effort slowly over the following decades (Figure 1b and c). Biomass levels have been increasing, especially after 2000 (Figure 1c and d), during which longline catch has increased as trawl and gillnet catches have declined (Figure 1d, ICES, 2015). RTCs have been implemented since the late 1960s, but the longline closures have increased as trawl closures have decreased as a reflection of recent gear shifts (Figure 1a and d). RTCs are currently triggered by catch numbers consisting of 25% or greater cod that are <55 cm.

Haddock

Landings of Icelandic haddock are currently low (Figure 2d) due to excessive fishing mortality and low survival have caused decreases in the stock until 2011. Since then, reduced fishing mortality has helped to slow decreases (ICES, 2015). Similar to cod, longline catch has increased while trawl and gillnet catches have decreased (Figure 2d, ICES, 2015). As a result, longline RTCs have increased while trawl RTCs have decreased (Figure 2a). Haddock RTCs are triggered by catches composed of more than 30% of haddock <45 cm.

Herring

Prior to 2008 fishing mortalities were steadily higher than target fishing mortalities, but high stock levels persisted until an *Ichthyophonous* infection reduced stock levels during the following 2 years (Figure 3, ICES, 2015). Most catch originates from pelagic trawls or purse seines (ICES, 2015). Other regulations are in place to reduce selectivity of herring, including certain gear and bycatch limitations, as well as RTCs that have been implemented since the 1970s (ICES, 2015). RTCs are currently triggered when >25% of a catch is composed of herring <27 cm.

Saithe

The harvest rate for saithe peaked in the 1990s, but since then has stabilized around a similar rate as intended by a newly introduced

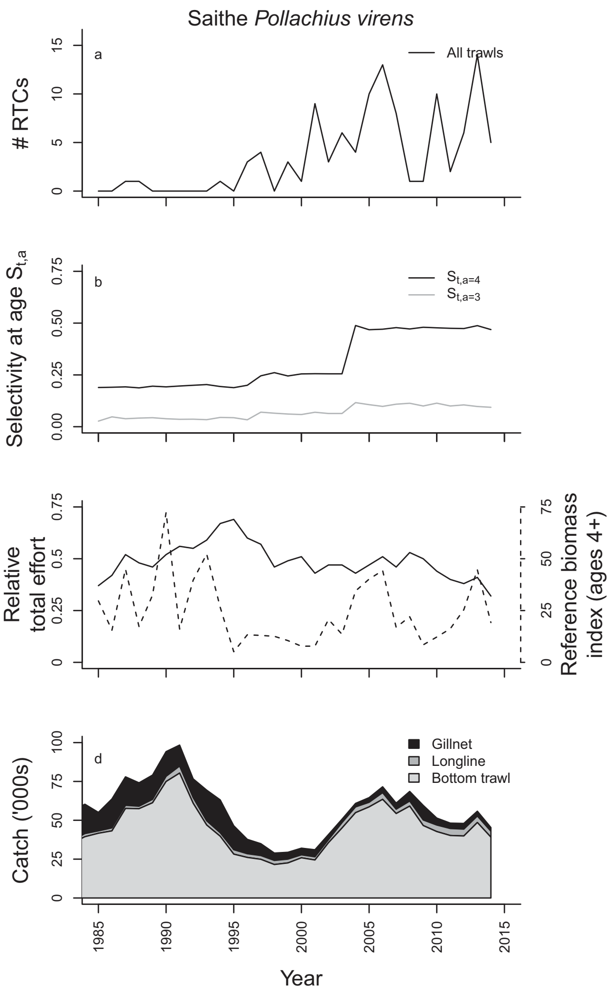

Figure 4. Time series of numbers of RTCs by gear (a), selectivity estimates for ages of recruitment (3) and entry (4) (b), relative total efforts (left axis) and reference biomass indices (right dotted axis) (c), and catch numbers by gear (d) are shown for saithe.

harvest control rule (20% of the reference biomass, ICES, 2015). Supporting trends can be seen in effort and catch data (Figure 4b and d). Saithe selectivity estimates are fixed as constant in the stock assessment within three periods: one occurring before 1996, from 1997 to 2003, from 2004 to present, leading to artificially large changes between periods (Figure 4b). Therefore, we did not analyse saithe selectivity estimates, only catch. We also included a dummy variable to reflect changes in catch from the periods 1990–2003 to 2004–2014 as a process unrelated to RTCs, as the stock assessment model indicates a large increase in selectivity around this time (Figure 4b). Small-sized saithe have recently been targeted to a greater extent than in the past, partially as a result of the reduction of gillnets that target larger fish (ICES, 2015). RTCs are triggered when catch of saithe <55 cm compose at least 30% of a catch in numbers.

Data analyses

Detecting susceptibility of small fish to being caught

RTCs are designed in general to reduce susceptibility of small fish to being caught. Therefore, if RTCs are successfully implemented, there should be more RTCs in years when catches with large proportions of small fish are more frequent (positive relationship). However, if RTCs are furthermore effective at reducing the susceptibility of small fish to being caught (i.e. they are working as intended), then this relationship should reverse: there should be lower values of selectivity and catch for small fish in years when there are a high number of closures, relative to other years (negative relationship). We expect model results to be quite similar among selectivity and catch, but we use both because there are pros and cons for using each source as data. Although changes in selectivity are ultimately what we are trying to detect, selectivity estimates come from stock assessment outputs, which means that their accuracies rely on model assumptions and any restrictions put on the model. For example, saithe selectivities were constrained to be constant within three time periods in the saithe stock assessment, so these were excluded from analyses. On the other hand, cod selectivities were estimated as random effects in statistical catch-at-age models, so although they are not true data, they are thought to be reliable as they are not constrained to take a specified shape (e.g. a dome or sinusoidal shape). Herring and haddock selectivities were estimated in a statistical virtual population analysis (VPA). Although some formulations of statistical VPAs can impose an asymptotic selectivity curve, the oldest age class selectivity values are generally the ones that are constrained, while younger age classes, such as those analysed here, should be less affected (Butterworth and Rademeyer, 2008). In contrast, catch-at-age data are not stock assessment output, despite being extrapolated from total catches and sub-sampled age distributions. However, catch is positively related to fishing effort and biomass of that age class, so it may only be used as an indicator of selectivity if changes if effort and biomass are accounted for. Effort was included as a covariate in main data analyses (see following section), whereas the effect of biomass of the same age class was removed *a priori* by first regressing catch over the same-age biomass index. Catch residuals ($\dot{C}_{t,a,s}$) were then used as the dependent variable instead of raw catch to reflect catch level relative to stock status.

Statistical framework

Four further statistical issues were apparent from the outset. First, although RTCs are meant to reduce susceptibility of recruits and younger fish to being caught, precision of selectivity estimates from recruitment-aged fish, indicated in stock assessments, are quite low due to generally low mortality rates relative to other age classes. Therefore, we focused on predicting selectivity and catch of the age class above recruitment, which we refer to as "entry-aged" and are generally the first year of inclusion in reference biomass (i.e. age 4 for cod, saithe, and herring, or 3 for haddock). In general, entry-aged selectivity estimates had higher precision than recruitment-aged estimates and were correlated with recruitment-aged estimates due to overlapping size ranges between recruit and entry-aged fish (Figures 1b–4b). As a result, there is likely to be a direct effect of RTCs on this age class as well.

Second, throughout the history of any fishery, stock biomasses and fishing effort levels will fluctuate in response to environmental or economic changes. Such fluctuations can affect the regulatory environment or stock age structure, as well as the behaviour of fishermen (e.g. through changes in gear or location), and consequently can also affect observed selectivity patterns. It is therefore useful to include variables reflecting the general state of the fishery to avoid confounding these changes with the effect of RTC closures. We included both relative total fishing effort and reference biomass in the analysis as covariates for this purpose.

Third, as these are time series, autocorrelation may need to be accounted for. To do this, we fit ARIMA time-series models with a matrix of external regressors (also known as ARIMAX models, Equations 1–3), and searched for the best combination of external regressors and autocorrelation parameters to explain patterns in selectivity or catch. This search included all possible combinations of predictors (i.e. the number of trawl, long-line, or herring-gear RTCs, and the covariates relative total effort, reference biomass, and a dummy variable for saithe reflecting a change in conditions from 2003 to 2004) and their interactions with the constraint that main effects were included when interactions composed of them were included. Both trawl and long-line RTCs were included for cod and haddock, only trawl RTCs were included for saithe, and only herring-gear RTCs were included for herring.

All analyses were restricted to 1990–2014, which coincides with the broad establishment of the ITQ system (Arnason, 2005). All variables were log-transformed to increase normality except effort variables. Unity was added to guard against zeros causing failure of definition.

ARIMAX model selection

For each possible combination of predictors, all possible combinations of 5 or fewer auto-regressive (AR_p, $\max(p) = 5$) or moving average (MA_q, $\max(q) = 5$) parameters were tested, as well as the order of differencing (d) needed to obtain independent, stationary residuals. For cod, we restricted the search to include only $d = 1$, as preliminary analyses indicated that single-lag differencing could avoid the inclusion of a dummy variable used to reflect a large change in selectivity due to the introduction of the harvest control rule (Figure 1), which would extend the exhaustive best-model search to an infeasible level. AR and MA parameters quantify autocorrelation within the error of an initial regression (Equations 1 and 2), with random error quantified as normal and independently distributed (Equation 3). In such a regression, the predictions at time t ($y_t^{'}$) are determined by a set of k predictors ($x_{t,k}^{'}$), their regression coefficients ($\beta_0 \ldots \beta_k$), and autocorrelated error ($n_t^{'}$). The prime notation ($y_t^{'}$, $x_t^{'}$, $n_t^{'}$) indicates that model fitting may occur on data that were differenced, the order d of which reflects how many times single-lag differencing was needed for residuals to become stationary according to a Kwiatkowski–Phillips–Schmidt–Shin (KPSS) test. For example, the single-order differencing used for cod ($d = 1$) indicates $y_t^{'} = y_t - y_{t-1}$ (Hyndman and Athanasopoulos, 2013). Model fitting was implemented in R (R Core Team, 2015), mainly using the "auto.arima" function in the "forecast" package (Hyndman and Khandakar, 2008). The best model was chosen as that which minimized Aikaike Information Criteria corrected for small sample sizes (AICc). The maximum and minimum residual from final models were tested as outliers using a Grubbs test in R package "outliers" (Komsta, 2011), which tests the difference between the residual and the mean over all residuals, divided by the standard deviation, against a t-distribution. The statistic U is the ratio of residual variances with and without the tested residual included. We use null and residual deviances, standard outputs of generalized linear models but not ARIMA models, to report a convenient standard measure of model fit assuming normally distributed error: percent variation explained ([null deviance−residual deviance]/null deviance). We also show t-statistics (coefficients divided by their standard errors), as tests indicate a difference from 0, and corresponding P-values with degrees of freedom ($n - d - p$ where n is the number of years of data and p is the total of linear and autocorrelation coefficients).

$$y_t^{'} = \beta_0 + \beta_1 x_{1,t}^{'} + \ldots + \beta_k x_{k,t}^{'} + n_t^{'} \tag{1}$$

$$n_t^{'} = c + AR_1 n_{t-1}^{'} + \ldots + AR_p n_{t-p}^{'} + MA_1 e_{t-1} + \ldots + MA_q e_{t-q} + e_t \tag{2}$$

$$e_t \sim N(0, \sigma^2) \tag{3}$$

Spurious correlations in time series

The last statistical issue noticeable from the outset was that, even after including covariates that reflect the general state of the fishery, any relationship we find between the number of RTCs and selectivity or catch could be due to spurious correlations with historical unobserved effects. For example, if observer coverage is correlated with the number of RTCs, perhaps greater observer coverage is the main mechanism reducing small-fish susceptibility to being caught rather than RTCs themselves. To rule out as many spurious correlations as possible, we repeat the presented analysis in Supplementary Material Section S2, but after removing the effects of a number of additional effects from the predictor (number of RTCs) before using it as a predictor in main analyses. In particular, we removed the effects of annual variation in observer coverage, recruitment, and gear-specific effort on number of RTCs, and correlations among number of RTCs from different gear types. However, approaching the problem as we did in Supplementary Material did not produce sufficiently different model results to justify presenting the more complex results here. We instead highlighted the main differences in "Results" section.

Results

Detecting susceptibility of small fish to being caught

For all species, entry-aged catch was regressed over entry-aged biomass with positive effects, so catch was adjusted for biomass levels by taking residuals (Table 1).

ARIMAX model selection

Using our model selection method, we found two practical problems. First, because the level of differencing is chosen after selecting the set of regression variables (using "auto.arima"), a peculiar situation can arise in which the level of differencing varies depending on the regression variables included. Every time a dataset is differenced, the time series is shortened by one data point, so that AICc values are no longer comparable: the shorter dataset automatically decreases the magnitude of the log likelihood. In these situations, there may be incomparable competing models at different levels of differencing. We chose to report the best model as that with more differencing (i.e. fewer data points), as these can be thought of as more conservative: by differencing the data, stationarity is removed *a priori* rather than as a result of

Table 1. Regression results from best models chosen by AICc to predict entry-aged catch numbers ($C_{t,a,s}$) using entry-aged biomass ($B_{t,a,s}$).

Species (s)	Predictors	Estimate	S.E.	t	P	% Expl.	Null df	Res. df	Residual formed
Cod	Intercept	0.0000	0.1697	0.0000	1.0000	31.00	24	23	$\dot{C}_{t,a=4,cod}$
	$B_{t,4,cod}$	0.5568	0.1732	3.2145	0.0038				
Haddock	Intercept	0.0000	0.1108	0.0000	1.0000	70.58	24	23	$\dot{C}_{t,a=3,haddock}$
	$B_{t,3,haddock}$	0.8401	0.1131	7.4284	<0.0001				
Herring	Intercept	0.0000	0.1704	0.0000	1.0000	30.41	24	23	$\dot{C}_{t,a=4,herring}$
	$B_{t,4,herring}$	0.5514	0.1739	3.1702	0.0043				
Saithe	Intercept	0.0000	0.1674	0.0000	1.0000	32.88	24	23	$\dot{C}_{t,a=4,saithe}$
	$B_{t,4,saithe}$	0.5734	0.1708	3.3568	0.0027				

Model parameter estimates, standard errors (S.E.), *t*-statistics (*t*), *P*-values (*p*), percent variation explained as calculated from null and residual deviances (% Expl.), and deviance degrees of freedom (df). Residuals formed ($\dot{C}_{t,a,s}$) were predicted in ARIMA models.

Table 2. Results of the best ARIMA models chosen by AICc to predict entry-aged selectivity ($S_{t,a,s}$) or catch residuals ($\dot{C}_{t,a,s}$) by each species *s* using (1) relative effort ($E_{t,s}$), (2) number of gear-specific RTCs ($R_{t,s,g}$), and (3) reference biomasses summed over appropriate age ranges ($B_{t,a,s}$).

Species	Dependent variable	d	Predictors	Estimate	S.E.	t	P	% Expl.	Null df	Res. df
Cod	$S_{t,a=4,cod}$	1	AR_1	0.4342	0.1877	2.3130	0.0309	40.05	23	21
			$R_{t,cod,trawl}$	0.1955	0.0876	2.2328	0.0366			
			$E_{t,cod}$	−0.5224	0.1770	−2.9509	0.0076			
	$\dot{C}_{t,a=4,cod}$	1	AR_1	−0.8515	0.1749	−4.8696	<0.0001	64.49	23	20
			AR_2	−0.4923	0.1743	−2.8240	0.0105			
			$R_{t,cod,trawl}$	0.4404	0.0905	4.8642	<0.0001			
			$B_{t,4+,cod}$	−0.2425	0.1013	−2.3950	0.0265			
Haddock	$S_{t,a=3,haddock}$	0	$R_{t,haddock,longline}$	−0.4170	0.1489	−2.8003	0.0102	50.00	24	23
			$B_{t,3+,haddock}$	−0.4551	0.1489	−3.0562	0.0056			
	$\dot{C}_{t,a=3,haddock}$	0	$R_{t,haddock,longline}$	0.2602	0.1296	2.0086	0.0576	73.79	24	21
			$E_{t,haddock}$	0.4303	0.1048	4.1053	0.0005			
			$B_{t,3+,haddock}$	−0.4288	0.1128	−3.8020	0.0010			
			$R_{t,haddock,longline} \times E_{t,haddock}$	−0.5593	0.1348	−4.1502	0.0005			
Herring	$S_{t,a=4,herring}$	0	$R_{t,herring,herring\text{-}gear}$	0.3534	0.1283	2.7548	0.0116	65.84	23	21
			$E_{t,herring}$	−0.5432	0.1180	−4.6029	0.0001			
			$R_{t,herring,herring\text{-}gear} \times E_{t,herring}$	−0.3784	0.1328	−2.8500	0.0093			
	$\dot{C}_{t,a=4,herring}$	0	$R_{t,herring,herring\text{-}gear}$	0.4769	0.1342	3.5550	0.0019	65.96	24	21
			$E_{t,herring}$	0.3963	0.1307	3.0327	0.0063			
			$B_{t,4+,herring}$	−0.2905	0.1374	−2.1143	0.0019			
			$R_{t,herring,herring\text{-}gear} \times E_{t,herring}$	−0.4283	0.1394	−3.0722	0.0063			
Saithe	$\dot{C}_{t,a=4,saithe}$	1	AR	−0.4308	0.1851	−2.3273	0.0295	24.80	23	22
			$E_{t,saithe}$	0.4911	0.2107	2.3306	0.0293			

Both trawl and longline RTCs were tested for haddock and cod; only herring-gear RTCs were tested for herring, and only trawl RTCs were tested for saithe. Autoregression parameters (AR), moving average parameters (MA), and the number of single-lag differences (d) are also shown, as well as linear model parameter estimates, standard errors (S.E.), *t*-statistics (*t*), *P*-values (*P*), percent variation explained as calculated from null and residual deviances (% Expl.), and deviance degrees of freedom (df).

correlations with regression parameters. However, in practice, competing models with different levels of differencing had very similar results.

Second, our model selection method also often tended to select overfit models. Therefore, we added selection criteria to exclude overfit models: the best model must include fewer than seven parameters, including autocorrelation parameters. In practice, the best models left after applying this criterion included far fewer parameters (Table 2).

For all models, we interpret only statistically significant results as the presence of an effect. For herring, 2013 selectivity was removed due to a positive outlier test showing extremely low residual selectivity that year in a preliminary model ($U = 0.6545$, $P = 0.0251$). All species with models predicting entry-aged catch had a positive effect (haddock, saithe, herring, $P \leq 0.05$) or no effect (cod, $P > 0.05$) of relative total effort (Figure 5b, d, f, and h, Table 2). These results are unsurprising because high effort levels lead to greater catch in general. In contrast, models predicting selectivity included a negative effect of relative total effort (cod and herring, $P \leq 0.05$, Figure 5a and e, Table 2) or no effect (haddock, $P > 0.05$, Figure 5c), indicating that years of high relative total effort were generally more likely to produce low susceptibility of small fish to being caught. Surprisingly, although biomass of the same age class was accounted for *a priori*, there was also a negative effect of reference biomass in models of cod, haddock, and herring catch, as well as haddock selectivity. High biomass years therefore also led to generally low catches of entry-aged fish. Results indicated that using ARIMAX models were helpful to counter some nuisances in fitting time series data, such as dealing with non-stationarity through differencing and accounting for

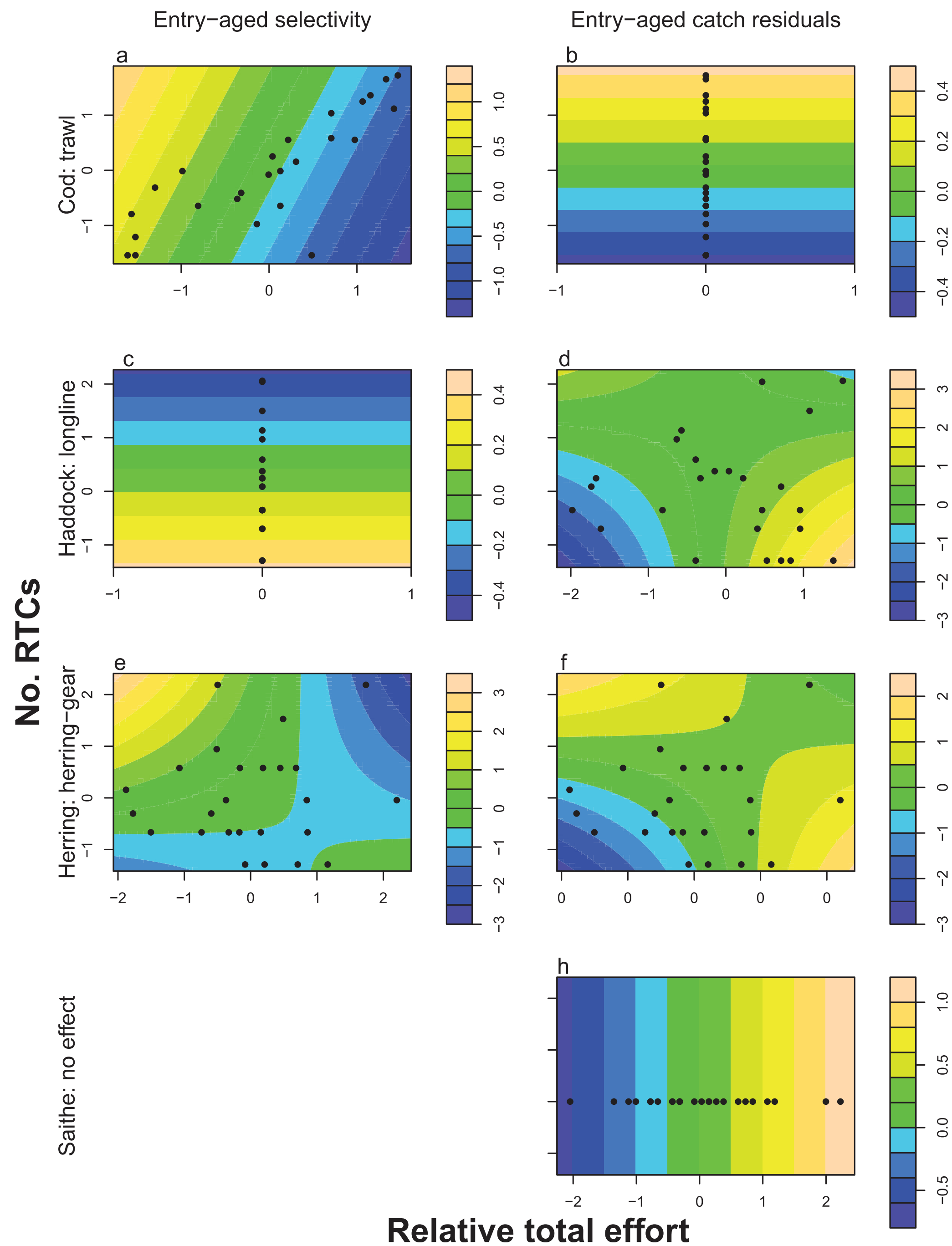

Figure 5. Contours represent predictions of entry-aged selectivity estimates and catch residuals resulting from the best-estimated ARIMA models presented in Supplementary Table S2. Main effects of predictors (excluding dummy variables and reference biomass) can be seen as linear changes in the prediction surface along the *y*-axis (the number of RTCs) or *x*-axis (relative total effort). Biomass was held constant at the mean value. Vertical gradients from dark colors at high values of the *y*-axis toward light colors at the low values of the *y*-axis correspond with our expectation, that selectivity is lower in years with more RTCs. The nonlinear contours found in panels d–f reflect effects of interactions, resulting in saddle shapes with areas of low selectivity estimates and catch values in upper right and lower left panel corners. Black points show the data used to estimate models. If the number of RTCs was not included as an effect, data were plotted along a horizontal 0 line, whereas data were plotted along a vertical 0 line if relative total effort was not included as an effect.

autocorrelation in error by including autoregressive or moving average parameters (Table 2).

The number of trawl RTCs (but not longline RTCs) was positively correlated with the susceptibility of small cod to being caught (i.e. selectivity estimates, Figure 5a) as well as cod catch (Figure 5b, Table 2). In contrast, the number of longline RTCs was negatively correlated with susceptibility of small haddock (Figure 5c), and had a negative interaction with relative total effort in the model of haddock catch (Figure 5d). This interaction resulted in a saddle-shaped contour, and indicated a reduction in small-haddock catch with greater numbers of longline RTCs only in years with relatively high effort (Figure 5d). Although the number of trawl RTCs was not selected in the haddock models as a significant effect, analyses in Supplementary Material show it to be correlated with longline RTCs (Supplementary Table S3). In both herring models, the inclusion of a positive main effect of the number of herring-gear RTCs alongside a negative interaction with effort caused a similar saddle shape as that found for haddock catch. Both the susceptibility of small herring to being caught and catch levels were reduced in years with a high number of RTCs, but only under high effort levels (Figure 5e). No effect of the number of RTCs on catch was found for saithe (Figure 5h).

Residuals of all ARIMA models presented here were generally normally distributed according to QQ and residual plots (Supplementary Figures S1 and S2 in Supplementary Material Section S1.2). No extreme residuals were found to be outliers using a Grubbs test. Autocorrelations and partial autocorrelations calculated from ARIMA residuals also did not differ from zero at any lag (Supplementary Figures S3 and S4).

Spurious correlations in time series

Results differed slightly from those presented in Supplementary Material where it was attempted to control for spurious correlations by removing effects of covariates from the number of RTCs *a priori* (Supplementary Material Section S2). The major differences shown in model results given in Supplementary Material include (1) an interaction between effort and the number of RTCs when predicting haddock selectivity, resulting in similar saddle shape as the haddock catch model presented here, (2) a three-way interaction for herring catch among the number of RTCs, relative total effort, and reference biomass, rather than the saddle-shaped effect presented here, and (3) a reduction in significance of RTCs on cod. The difference in herring results is attributed to the removal of a strong positive effect of recruitment on the number of herring-gear RTCs in Supplementary Material analyses. The difference in cod results appears to be related to the removal of a positive contribution of trawl-specific effort or recruitment to observed cod selectivity and catch (Supplementary Table S2). Qualitatively, model interpretations are similar to those presented here.

Discussion

Here we analysed whether an effect of the Icelandic RTC system could be detected in either selectivity estimates, reflecting the susceptibility of small fish to being caught, or catch levels, relative to biomass, of young fish over time. An increase in selectivity estimates or entry-aged catch with number of closures would indicate greater fishing mortality, relative to other age classes, in years when RTCs are triggered more often. Such a positive effect therefore would affirm that the RTC system was being triggered as expected: more closures occur in years when more entry-aged fish are being caught in relation to other age classes. However, positive effects do not lend credence to the idea that the fishery closures are ecologically effective; that is, that they reduce susceptibility of small fish to being caught. To have the desired effect on population dynamics, any positive correlation must be overwhelmed by negative effect of closures on susceptibility to being caught; that is, RTCs should cause a reduction in fishing mortality, and possibly catch as well, of small fish.

Although the entire RTC system includes far more species than the four we analyse in this study (Schopka, 2007), we restricted the analysis to species with more reliable stock assessments that could yield selectivity estimates, and biomass estimates used to adjust catch data. We found that two of the four species, haddock and herring, showed evidence of ecological effectiveness through a reduction in selectivity estimates and catch of entry-aged fish in years with greater numbers of RTCs. However, this effect was only detectable as an interaction with effort in both herring models and the haddock catch model, indicating that the reduction in small-fish susceptibility and catch is only detectable in high-effort years. Cod, on the other hand, only showed a positive relationship, affirming that the RTCs were being triggered properly. No effect was detected for saithe.

We can only speculate why certain species exhibit effects of the RTC system while others do not. There are a variety of potential biological mechanisms that can lead to differences in how effective an RTC system may be in reducing mortality. How aggregated a species age class is, or whether it schools, can cause a species age class to be more or less susceptible to fishing gear, and therefore how sensitive selectivity patterns are to slightly different fishing practices. Similarly, if a species is more likely to mix with other species or age classes, the probability of initiating triggers may differ among species, and their effectiveness may be reduced. As herring are pelagic and form highly aggregated size-specific schools, it is not surprising that they are one of the two species with effective RTCs. However, of the species analysed here saithe is the second-most prone to strong size-specific schooling behaviour and spends more time in the water column than the other gadoids (Scott and Scott, 1988; Neilson *et al.*, 2002, 2003), but did not show an effect of RTCs. Perhaps because saithe are also fast swimmers, patchily distributed, and generally elusive to find (Pol *et al.*, 2016), aggregations may exit a closed area before any protection against mortality can be afforded. Haddock and cod are more similar as they spend more time searching for food close to the seabed rather than in the water column (Anders *et al.*, 2017). However, RTCs were only ecologically effective for haddock, not cod. Perhaps the system is not effective enough to reverse the cod selectivity and catch trends, supporting previous suppositions made based on preliminary analyses that the RTC system likely has no effect on cod dynamics (ICES, 2011; Needle and Catarino, 2011).

Besides differences in ecology, differences in the behaviour of fishers targeting each species could likewise affect RTC system effectiveness. For example, fishers may already actively avoid small sizes of some species so that the system becomes redundant. Conversely, if smaller fish fetch a higher price or are otherwise desirable, then fishers have an incentive to target small sizes, thereby counterbalancing any reductions in selectivity yielded by the RTC system. For example, if the closures are not of a sufficient length, movement of fishers back into closed areas can negate any effect of the closures (Needle and Catarino, 2011). Alternatively, fishers could avoid triggering the RTCs while observers are on board by fishing in other locations. For saithe, recent years are marked by a rise in selectivity of small fish (Figure 4b), possibly due to direct targeting (Condie *et al.*, 2014), but this is difficult to distinguish from unrelated changes in gear (i.e. a reduction in gillnet usage, Figure 4d). Furthermore, concurrent regulations may also reduce the incentive to avoid small-sized fish at the same time as RTCs increase it. For example, Condie *et al.* (2014) point out that despite the presence of a discard ban and RTC system in Iceland, there is very little evidence to suggest that these regulations have changed selectivity patterns of fishers, especially as discards still occur and there are still a large number of small-sized fish landed under a 5% overquota provision. In addition, mandatory sorting grid regulations were implemented in 2000 but reduced in 2006 and 2013, thereby changing selectivity patterns both directly and by making certain fishing locations more accessible.

Despite an inability to pinpoint causes for differing results among species, this study at least serves as proof-of-concept: it is the first to show that RTCs aimed at reducing susceptibility of small-sized fish to being caught are likely effective for some species. Haddock and herring, both of which are highly profitable fisheries in Iceland, exhibited reduced selectivity of small fish under high effort conditions, and therefore likely produce economic benefits, given that changes in selectivity on small fish will cause increased yield at older ages. Although Supplementary Material analyses indicate that the effect of RTCs on small herring is confounded with herring recruitment, we are not aware of any other specific mechanism that would reduce small-herring susceptibility in high-recruitment years, so it would be premature to attribute herring results to a spurious correlation. Nonetheless, this study also suggests there is room for improvement. For example, the closures are for the same length of time for all species, even though life histories are quite varied. Although species-specific, it may be possible to refine rules for triggering RTCs using data on movement and site fidelity (Dunn *et al.*, 2014). Similarly, efficiency may be improved through analyses of catch composition data to optimally design trigger thresholds or fine-tune closures according to time, place, or environmental conditions. Finally, if the greatest benefits are reaped when fishing effort is high, then effort or overall fishing mortality could theoretically be incorporated into the trigger. In this case, analysing the relationship between selectivity patterns and effort levels, as well as how they differ by gear or change through fishing seasons, should be given priority for assessing when and how RTCs could be made most effective. This would be a good starting point, for example, to understand why RTCs appear not to be effective for cod, followed by analyses regarding other regulations or fishing behaviour that may counter-balance RTCs. Given that cod dominates the demersal fisheries in Iceland by far, potentially large gains could be made by small modifications to the current system. A cost-benefit analysis would be a logical next step in evaluating by how current implementation of RTCs can be improved (e.g. through move-on rules, Dunn *et al.*, 2016).

While examining the benefits of RTCs, two additional non-monetary benefits should likely be considered. First, a greater awareness among both the public and industry participants can develop regarding the importance of avoiding recruitment overfishing, as closures are advertised widely in public media when triggered. In conjunction with a discard ban, size-based closures can incentivize technological innovation and voluntary use of selective fishing gear, as well as promote the idea that discarding is wasteful and morally wrong. This was the experience in Norway when implementing RTCs in the 1980s in the Norwegian and Barents Sea cod and haddock fishery (Graham *et al.*, 2007; Condie *et al.*, 2014; Gullestad *et al.*, 2015). In Iceland, fishing captains can voluntarily trigger a closure by self-reporting (as verified by three independent fishing captains). Sharing responsibility or co-designing and co-managing the system with industry partners can also increase a sense of stewardship and communication among industry participants (O'Keefe and DeCelles, 2013). Often, motivations for co-management, gear innovations, or innovative pilot projects have arisen out of an impending regulations that threaten to reduce fishery profitability, such as proposed permanent closures (e.g. see Needle and Catarino, 2011; Bethoney et al., 2013; Condie *et al.*, 2014; Eliasen, 2014; O'Keefe *et al.*, 2014; Lewison *et al.*, 2015; Little *et al.*, 2015). Real-time spatial management can therefore reduce conflict between management and industry by minimizing regulatory burden and the mismatch between management goals and economic outcomes, thereby potentially increasing trust and compliance (Kraak *et al.*, 2015; Maxwell *et al.*, 2015; Dunn *et al.*, 2016).

Second, although reductions in selectivity may be possible through more permanent time-area closures (Dunn *et al.*, 2011), as Eliasen (2014) suggests, in many cases the modelling necessary to justify and define such closures requires a longer time series of data and cannot yet be accomplished, in part because permanent closures are often not supported by the industry and become politically contentious issue (O'Keefe *et al.*, 2014; Lewison *et al.*, 2015; Dunn *et al.*, 2016). In the meantime, RTCs can allow for immediate response to a known, but incompletely defined problem. As is the experience in Iceland, RTCs can also aid the development of more permanent closures by gradually accumulating necessary data (Schopka, 2007; ICES, 2011). Finally, this study points out the weaknesses of analysing fisheries regulations without an experimental design. Therefore, if there is a desire to implement RTCs in other locations, then there is also an opportunity to design its implementation such that causal effects of the regulation may be more rigorously detected.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

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