



Original Article

Effects of time-area closures on the distribution of snow crab fishing effort with respect to entanglement threat to North Atlantic right whales

Alexandra K. Cole ^{1,2,*}, Sean W. Brillant ^{1,3}, and Stephanie A. Boudreau ⁴

¹Canadian Wildlife Federation, 350 Michael Cowpland Drive, Kanata, ON K2M 2W1, Canada

²Marine Affairs Program, Dalhousie University, Life Sciences Centre, Biology Room 805, 1355 Oxford Street, PO Box 15000, Halifax NS B3H 4R2, Canada

³Department of Oceanography, Dalhousie University, Life Science Centre, Room 3631, 1355 Oxford Street, PO Box 15000, Halifax NS B3H 4R2, Canada

⁴Department of Fisheries and Oceans Canada, Gulf Fisheries Centre, 343 Université Avenue, P.O. Box 5030, Moncton N.B. E1C 9B6, Canada

*Corresponding author: tel: +1-877-599-5777 ext. 205; e-mail: alexandrac@cwf-fcf.org

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Time-area closures are increasingly used to mitigate cetacean entanglement by temporarily excluding fishing effort from areas where high densities of cetaceans and fishing overlap. The effort displaced by these closures can be redistributed to the areas that remain open, changing the distribution and density of fishing effort outside the closures. These patterns were evaluated for the southern Gulf of St. Lawrence snow crab fishery by comparing recent years (2015–2017) with 2018 when time-area closures were implemented to protect North Atlantic right whales. A predictive model framework was created to test how well we could predict the response of fishers to closures. Approximately 29% of the total fishing effort was displaced by the 2018 closures, increasing effort density outside the closures by 41%. Displaced fishing effort shifted farther from the closures than predicted, into areas which, prior to 2018, had low effort density, producing a higher threat of entanglement in these new areas. Fishing effort in 2018 remained as high as 2017, despite a lower quota and reduced trap limit. Consequently, the resulting effects of time-area closures on fishing patterns outside of the closures cannot be discounted if entanglement threat to whales is to be successfully mitigated.

Keywords: bycatch, cetaceans, conservation management, entanglement, fisheries displacement model, fisheries management, fishery closures, Gulf of St. Lawrence, spatio-temporal closure

Introduction

Entanglement in commercial fishing gear is one of the leading causes of cetacean mortality and injury worldwide (Read, 2008; Cassoff *et al.*, 2011; Moore and van der Hoop, 2012; van der Hoop *et al.* 2012, 2014, 2017; Knowlton *et al.*, 2016; Sharp *et al.*, 2019). Entanglement occurs when a cetacean encounters fishing gear and it becomes wrapped around parts of its body. Thus, entanglement risk is a function of the densities of fishing effort and whales in a given area, at a given time (Johnson *et al.*, 2005; Vanderlaan *et al.*, 2011; Brillant *et al.*, 2017; DFO, 2019c). Time-area closures are a

management measure intended to address this issue of overlapping distributions, fulfilling both fisheries and conservation objectives by excluding potentially harmful fishing activity from areas where at-risk species occur, while allowing fishing to continue normally outside of the closures (Hall, 2002; Lewison *et al.*, 2004; Dunn *et al.*, 2011; Dichmont *et al.*, 2013). As such, time-area closures are increasingly used globally to prevent cetacean entanglement (Murray *et al.*, 2000; Merrick *et al.*, 2001; Dawson and Slooten, 2005; Farmer *et al.*, 2016; Leaper and Calderan, 2017).

Entanglement risk in areas closed to fishing is expected to be reduced to zero (Murray *et al.*, 2000; Dawson and Slooten, 2005;

Johnson *et al.*, 2005). However, fishing effort that was located within the closure boundaries must relocate if fishing is to continue. Moreover, cetaceans regularly move in and out of protected areas, and complete habitat use is not always known, thus the displacement of fishing effort can increase the entanglement threat outside the closures, influencing the overall expected net reduction of entanglement risk (Murray *et al.*, 2000; Halpern *et al.*, 2004; Murawski *et al.*, 2005; Greenstreet *et al.*, 2009; Agardy *et al.*, 2011; O'Keefe *et al.*, 2014; Farmer *et al.*, 2016; Hoos *et al.*, 2019). Few studies have evaluated these potential counteracting effects of displaced fishing effort from time-area closures (Dinmore *et al.*, 2003; Hiddink *et al.*, 2006; Greenstreet *et al.*, 2009; Abbott and Haynie, 2012; Dichmont *et al.*, 2013; Hoos *et al.*, 2019), and even fewer as it relates to cetacean entanglement (Murray *et al.*, 2000; Farmer *et al.*, 2016). This knowledge is important to ensure the overall entanglement threat is sufficiently reduced, especially where there is an urgent need for successful conservation measures, such as with the critically endangered North Atlantic right whale (*Eubalaena glacialis*, hereafter right whale; Murawski *et al.*, 2005; O'Keefe *et al.*, 2014; Slijkerman and Tamis, 2015; Chollet *et al.*, 2016).

Right whales have long been one of the most endangered large whales, with a current estimated population of 356 individuals (Caswell *et al.*, 1999; Pace *et al.*, 2017; Pettis *et al.*, 2021). Ranging along the eastern United States, where they typically spend the fall to early spring, to Atlantic Canada in late spring through to fall, the majority of recorded right whale mortalities and injuries are attributed to entanglement (Knowlton *et al.*, 2012; van der Hoop *et al.*, 2012; Kraus *et al.*, 2016; Sharp *et al.*, 2019). Following a recent shift of increased right whale occurrences to the southern Gulf of St. Lawrence (sGSL), a mortality event occurred in 2017 (Davies and Brilliant, 2019). That year, 17 right whales died across their range, 12 of which were in the sGSL along with five other live-entanglements (Daoust *et al.*, 2017). At least two of these deaths and four of the live-entanglements were the result of encounters with Canadian snow crab (*Chionoecetes opilio*) fishing gear, which has a large overlap with the newly observed aggregations of right whales in the sGSL (Figure 1; Daoust *et al.*, 2017; Davies and Brilliant, 2019). This spurred the Government of Canada to implement new measures to prevent interactions between snow crab fishing gear and right whales (DFO, 2018; Davies and Brilliant, 2019).

Snow crab is Canada's second most lucrative fishery (2018 estimate: \$748 million CAD) with the sGSL region contributing approximately half the total annual landings and value (DFO, 2020). Snow crab is a quota-based fishery whose quota (total allowable catch; TAC) is divided amongst approximately 760 licence holders from Nova Scotia, New Brunswick, Quebec, and Prince Edward Island (PEI) across four crab fishing areas (CFA; Figure 2; DFO, 2014b, 2019a). Trap limits and individual quota allocations within each fleet vary annually based on management regulations (DFO, 2014b). In 2018, the maximum allowable traps per licence was 150, which was typical for recent years; though for the 2017 season, up to 174 traps were allowed due to an elevated annual recruitment of snow crab (DFO 2014b, 2017, 2018; Davies and Brilliant, 2019). Traps are baited and set individually on the ocean floor, then left unattended for at least 24 hours after which they are retrieved and immediately reset (Hébert *et al.*, 2020). Each trap is connected to a surface buoy via an attached rope (i.e. buoy line), and thus each trap, with its buoy line, represents an entanglement threat (Johnson *et al.*, 2005).

After emergency measures were implemented during the 2017 mortality event, seasonal management measures intended to miti-

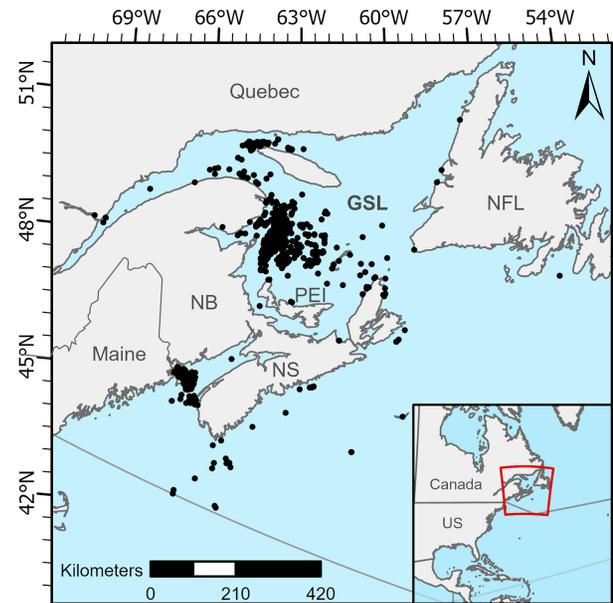


Figure 1. Canadian North Atlantic right whale sightings, uncorrected for survey effort, from 2015–2018 (Right whale Consortium, 2020).

gate entanglement risk to right whales were introduced for the 2018 sGSL snow crab fishing season (DFO, 2018; Davies and Brilliant, 2019). The measures focused primarily on the largest CFA, area 12 (Figure 2), typically fished from late-April (once ice conditions are safe) to mid-July (Hébert *et al.*, 2020). To minimize overlap of right whales and fishing activity, the 2018 season was shortened, ending on June 30, and two types of time-area closures were implemented: a fixed “static” fishing exclusion zone, and dynamic management areas (DMA; Figure 2). The closures were delineated using a pre-existing grid system used for a soft-shell crab monitoring protocol intended to protect newly moulted (soft-shell) snow crab from handling mortality (Hébert *et al.*, 2020). The static closure, designed to encompass 90% of the reported 2017 right whale sightings, came into effect on 28 April 2018 and remained for the duration of the fishing season; no fishing was permitted within this area. The DMAs were actively surveyed (aerial- and vessel-based) to record real-time sightings of right whales. Based on these surveys, if at least one right whale was sighted within a grid cell (hereafter, cell) located in a DMA, that cell and the surrounding cells (up to a total of nine) were closed for a minimum of 15 days. These dynamic closures were extended for an additional 15 days from the last confirmed sighting until a whale was no longer seen in the original cell by two aerial surveys within the 15-day closure period; closed cells would be extended as a precautionary approach until an aerial survey could be conducted. Once a dynamic closure was triggered no new fishing activity could occur in these cells and fishers were given a minimum of 48 hours to remove existing gear located within the closed cells.

We were interested in examining fishing effort displaced as a result of the time-area closures to evaluate how this changed the distribution of entanglement threat outside of these closures. We evaluated fishing effort as an estimate of the threat of entanglement (i.e. effort density), rather than the risk of entanglement (i.e. the probability of an entanglement to occur) as risk requires effort corrected densities of right whales (DFO, 2014a), which are not yet adequately representative for this area, limiting us from producing reliable es-

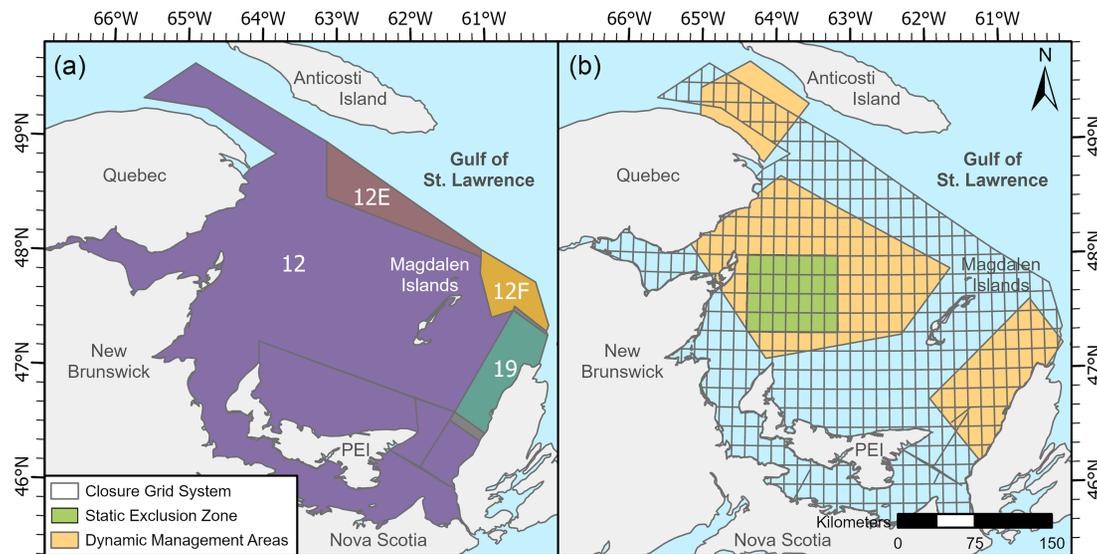


Figure 2. Southern Gulf of St. Lawrence snow crab fishing areas (a) and the management grids used for time-area and soft-shell crab closures (b). The static exclusion zone is in green and DMAs in orange.

timates of risk (Davies and Brillant, 2019). By comparing the 2018 sGSL snow crab fishing season to recent years prior to 2018, we evaluated: (i) if new areas of high effort density (i.e. threat) were created where effort density was previously low; and (ii) if the entanglement threat disproportionately increased along the perimeter of the time-area closures due to displaced fishing effort (Hall *et al.*, 2000; Murray *et al.*, 2000; Abbott and Haynie, 2012; Bastardie *et al.*, 2015; Chollett *et al.*, 2016; Hoos *et al.*, 2019). We also developed a model framework to attempt to predict the redistribution of fishing effort to evaluate how a fishery and the conservation benefits may change due to different time-area closure strategies to support effective management.

Methods

Estimates of fishing effort distribution

Fisheries and Oceans Canada (DFO) Gulf region provided complete, but anonymous, commercial snow crab fishing logbook data for CFA 12 for 2015 through 2018. Logbook data were compiled by DFO's Informatics and Statistics Quebec and Gulf Region branches and verified for valid fishing locations. The number of traps hauled, and landings were recorded by GPS location (latitude and longitude) and date. The data did not distinguish different fishing fleets.

The spatial management system was a grid consisting of 509, 10-minute cells each covering 222–240 km² across the entire sGSL snow crab fishery, including the cells that would subsequently become closed in 2018 (Figure 2; DFO, 2015; Hébert *et al.*, 2020). These cells were used as the unit of spatial scale, and a standard week (seven days) as the temporal unit for this study. To standardize our estimates, a 24-hour soak-time was assumed, though it may be longer (Hébert *et al.*, 2020), so the set date of each trap was established as the day prior to the date the gear was retrieved. Fishing effort was calculated as the sum of traps set within a cell for each week. As the same trap may be set, retrieved, and reset multiple times through a week, this measure of effort was referred to

as trap-sets to distinguish this from the absolute number of active traps. Landings (tonnes) were also summed by cell for each week. Due to the anonymous nature of the logbook data, estimates of fishing activity for each week were treated as separate events. Furthermore, for simplicity, dynamic closures (for whales or soft-shelled crab) established during a week were assumed to be in effect for that entire week regardless of the actual day the closure began or reopened (e.g. a cell closed on day seven was considered closed for the entire week).

The study period was 22 April–30 June (2015–2018) comprising 10 weeks that were individually examined for each year. Despite interannual variance of snow crab abundance and distribution, prior to establishing the time-area closures, the spatial distribution and relative effort of fishing activity compared to the annual TAC was reasonably stable and consistent from year to year (DFO, 2019b). Therefore, to estimate the average distribution of fishing activity in the years prior to the implementation of the closures, weekly fishing activity was averaged from 2015 through 2017 (hereafter, the pre-closure period). This average distribution was compared to the 2018 distribution (hereafter, the observed period) to identify changes in fishing effort (i.e. number of trap-sets) that resulted from the time-area closures. These changes in distribution were measured based on area (i.e. number and location of unique cells fished) and effort density (i.e. trap-sets per cell).

The amount of fishing effort displaced due to the 2018 closures was estimated as the average fishing effort of the pre-closure period occurring within each cell closed in 2018. Changes in weekly effort density for all cells were expressed as the percent difference in effort before and after closures were implemented

$$\Delta E_k = \frac{E_{Bk} - E_{Ak}}{E_{Ak}} \times 100, \quad (1)$$

where E_A was the summed effort in all fished cells after displacement caused by a closure (e.g. observed period) in week (k), and E_B is the summed effort in all fished cells before closures were implemented (i.e. pre-closure period) in week (k). This was used to

examine the change in effort between the pre-closure and observed periods, as well as the changes predicted by the model.

Finally, changes in weekly fishing effort along the perimeter of the closures were measured in two ways: (i) the proportion of the perimeter (i.e. number of boundary cells) that was fished during 2018 compared to the pre-closure period; and (ii) the effort density of each cell along the perimeter of the closed areas during 2018 compared to the pre-closure period. We referred to these two definitions collectively as the boundary threat to reflect the two methods of measuring changes in threat along the perimeter of closed areas. These estimates were expressed as proportions to account for the changing sizes of the closed areas among weeks.

Predicting fishing effort displacement

We developed a model framework to test how well we could predict the distribution of fishing effort resulting from time-area closures using the pre-closure period as the base distribution before displacement. For each week, we identified each cell that was closed in 2018, and the average fishing effort that occurred in those cells during the pre-closure period. If no fishing activity occurred in any of the closed cells, fishing was assumed to be unaffected by closures, therefore no effort was relocated. If fishing had occurred within those cells during the pre-closure period, this effort was considered to be affected by the closures and must be moved elsewhere (hereafter, displaced effort). Only fishing activity within closed cells was relocated; fishing outside the closures was not moved (Powers and Abear, 2009). To predict the destination cell (i.e. a cell that received displaced effort), we used a series of rules and assumptions to mimic fisher behaviour. To inform these assumptions, a small survey of fishers ($n = 16$) from the Acadian Crabbers Association (ACA) was completed to provide insight on decisions fishers made related to relocating fishing effort due to the 2018 time-area closures (Table S1; R. Haché, pers. comm.).

The first assumption was that fishers followed a modified ideal free distribution, where they were considered omniscient and effort moved among cells based on the suitability of a new location to maximize their return (i.e. catch per unit effort, CPUE) relative to the distance traveled (Gillis et al., 1993; Hutton et al., 2004; Powers and Abear, 2009). Second was that past fishing distributions reflected local knowledge of where successful catches were most likely. This approach was corroborated by results from the ACA survey, and similar approaches have been used in other studies (Table S1; Hiddink et al., 2006; Greenstreet et al., 2009; Hoos et al., 2019). Therefore, potential destination cells were only those that were actively fished during that specific week of the pre-closure period. Third, the CPUE of the destination cells did not change as a result of the addition of the displaced effort. There is a weakly positive correlation between CPUE and biomass for this fishery (Swain and Wade, 2003; Hébert et al., 2020), supporting this assumption that changes in effort do not affect the expected landings of a destination cell. Finally, it was assumed that all effort displaced from one cell must fit within a single destination cell (i.e. displaced trap-sets from a closed cell were not divided among different destination cells). This assumption was used to simplify the model calculations because we could not account for a smaller spatial scale as the data were anonymous.

Additionally, two factors were used to determine the suitability of a destination cell: the maximum distance fishers would travel to relocate displaced effort, and the maximum effort density within a destination cell (i.e. weekly summed trap-sets). Based on input

Table 1. Parameters of each suitability variation. Maximum distance is the furthest distance a fisher would travel to relocate displaced gear. Maximum effort density is the maximum number of weekly trap-sets per cell. If the summed trap-sets exceeded this number, a displaced fisher would opt to not place their traps in that cell.

Maximum effort density	Maximum distance	
	75 km	138 km
2006 trap-sets	Variation 1	Variation 2
1457 trap-sets	Variation 3	Variation 4

from the ACA survey (Table S1; R. Haché, pers. comm.), we tested two maximum distances, 75 and 138 km, representing the 50th and 95th percentile of responses. We also tested two maximum effort densities, 2006 and 1457 trap-sets per cell (i.e. the 95th percentile of the weekly effort density of all cells observed during 2018 and the 95th percentile of the total effort density of all cells observed during 2018, respectively). We evaluated both levels of these two parameters in combination to determine which variation produced the most accurate results compared to the 2018 distribution (i.e. the observed period; Table 1).

Using these assumptions, the process to determine the destination of displaced fishing effort consisted of three main steps (Figure 3). First, only fished cells within the maximum distance of the displaced cell (Table 1) were considered. Second, the return, Z , for each of these cells was calculated as

$$Z_{ijk} = (C_{jk} / E_{jk}) d_{ij}^{-1}, \quad (2)$$

where C is the total landings (tonnes) of a destination cell (j) for week (k), and E is effort as the total number of trap-sets in that cell (j) for week (k), and d was the centroid distance (km) between the displaced cell (i) and the destination cell (j). Finally, the maximum effort density for the destination cell could not be exceeded (Table 1). Thus, to determine the destination cell of a displaced cell, the ten destination cells with the highest return (Z) for the displaced cell were ranked. In rank order, the displaced effort was added to the effort in the destination cell, but if the resulting effort density exceeded the maximum effort density, that cell was not selected and the next ranked cell was considered. If a suitable destination could not be selected from among the ten options, the displaced effort was removed from the fishery (Figure 3). The total number of suitable destination cells varied for each displaced cell and where multiple displaced cells competed for the same destination cell, the displaced cell that was closest moved there first (Gillis, 2003).

To determine which parameter combination provided the most accurate results (i.e. most closely matched the 2018 observed distribution), Spearman's rank correlation tests were performed between the predicted and observed distributions for each week for each of the four variations. The variation with significant results ($P \leq 0.05$) and the largest averaged correlation coefficient (r) was selected as the best fit. To ensure comparability in this analysis, the observed data (i.e. the sum of the fishing effort for all cells) were scaled to the same total effort (trap-sets) as the pre-closure period.

A chi-square test was performed for all cells for each week between the scaled observed data and the data predicted by the chosen model variation to test if the predicted and observed effort densities were independent. Data from Week 1 were excluded from these

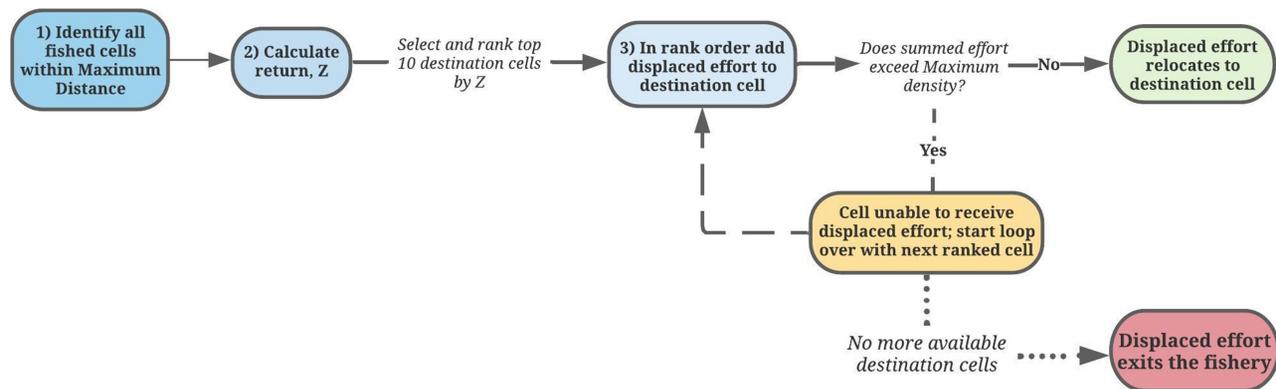


Figure 3. The framework for selecting destination cells for displaced fishing effort. Only cells within the maximum distance from the displaced cell were considered. The ten best destination cells by CPUE per km travelled (Z) were ranked, and in order the displaced effort was added to the existing effort of each destination cell. If the sum did not exceed the maximum effort density, the displaced effort was relocated to that cell. If maximum effort density was exceeded, the next ranked destination cell was considered. If displaced effort could not be relocated to any of these ten cells, it was removed from the fishery.

analyses as Week 1 of 2018 only had one day of fishing activity, so comparison with the pre-closure data was not appropriate.

Results

Shifts in fishing effort distribution

The 2018 static time-area closure consisted of 28 cells, and the number of unique cells affected by weekly right whale dynamic closures, which started in Week 5, varied from 6 to 36 cells over the 10-week season (Figure S1). This resulted in 64 unique cells (28 cells in the static closure and 36 cells as dynamic closures) being closed during the 2018 season. An additional four cells were closed due to soft-shell crab closures totalling 68 unique cells being closed by the end of the fishing season; 65 of these had been fished during the pre-closure period (average $88 \pm 2\%$ SE). The total number of unique cells fished during 2015, 2016, and 2017 were 146, 149, and 182, respectively, with a total of 195 unique cells fished over the entire 3-year pre-closure period. Between each year of the pre-closure period (2015 vs. 2016, 2016 vs. 2017, and 2015 vs. 2017), on average, 20% ($\pm 4\%$ SE) of fished cells were newly explored (i.e. fished in the current year but had not been fished previously) and 6% ($\pm 1\%$ SE) were abandoned (i.e. fished previously but not in the current year), annually. In 2018, over the season, 160 unique cells were fished, of which 24 (15%) were newly fished (i.e. not fished during the previous three years), while 29 (17%) which had been fished during the pre-closure period were not fished in 2018. None of these 29 were closed due to time-area closures. Thus, fewer new cells were explored; and more cells were abandoned in 2018 than the pre-closure average.

The displacement of fishing effort from the areas closed during 2018 increased weekly fishing effort density on average by 41% ($\pm 2\%$ SE; Equation 1) in the cells unaffected by closures. Moreover, much of this effort shifted away from previously high effort density areas near the closures to previously lower effort areas in the southeast sGSL, between PEI and the Magdalen Islands (Figures 4, S2–S5).

Fishing effort varied among years, but effort changed notably in 2018 relative to the TAC compared to the pre-closure period. In 2015 and 2016, CFA 12 had a TAC of 23021 and 19393 tonnes (DFO, 2019b), respectively, and a seasonal effort of 290277 and 272202

trap-sets, respectively, over our 10-week study period. In 2017, the TAC increased to 39651 tonnes (DFO, 2019b) with 456219 trap-sets during the 10-week period. In 2018, the TAC was reduced to 20909 tonnes, similar to 2015 and 2016 (DFO, 2019b), yet effort remained high and comparable to 2017 with 425501 trap-sets during the 10-week period (Figure 5). The average annual effort of the pre-closure period was 339556 trap-sets for the 10-week study period, of which 98122 (29%) were located within the 65 fished cells that would become closed in 2018 (Figure S6), representing the effort that was expected to be displaced during 2018.

The average proportion of cells surrounding the 2018 closed areas (i.e. boundary cells) that were fished during the pre-closure period was 53% ($\pm 3\%$ SE), but during 2018, once closures became active, only an average of 43% ($\pm 5\%$ SE) of these boundary cells were fished. While the proportion of total boundary cells fished decreased, the average effort density within the fished cells increased. During the pre-closure period, an average of 10% of the seasonal fishing effort occurred in these boundary cells compared to an average of 14% during 2018 (Figures 6, S7–S8).

Predicting fishing effort displacement

The Spearman's rank correlation tests for each of the four model variations were all significant ($P \leq 0.05$; Table S2) and had similar correlation coefficients (degree of correlation, r ; Table S2), but Variation 1 (75 Km, 2006 trap-sets) produced the largest average correlation coefficient and was thus chosen as the best-fit parameters. Chi-squared tests between the weekly predicted effort density distributions using the best-fit parameters and observed effort density distributions showed the framework did not adequately predict the displaced distribution, as non-independence between these distributions was only rejected for one of the nine tested weeks ($P \leq 0.05$; Table S3).

The model overestimated the amount of fishing effort displaced to cells adjacent to the time-area closures (i.e. boundary cells). Over the 10-week period, the model predicted 71448 (80%) of the total displaced trap-sets would relocate to boundary cells and produce an overall 237% average ($\pm 28\%$ SE) increase in effort in these cells. This was higher than the observed average overall effort density increase of 44% ($\pm 7\%$ SE; Figures 7, S9–S23) in these boundary cells.

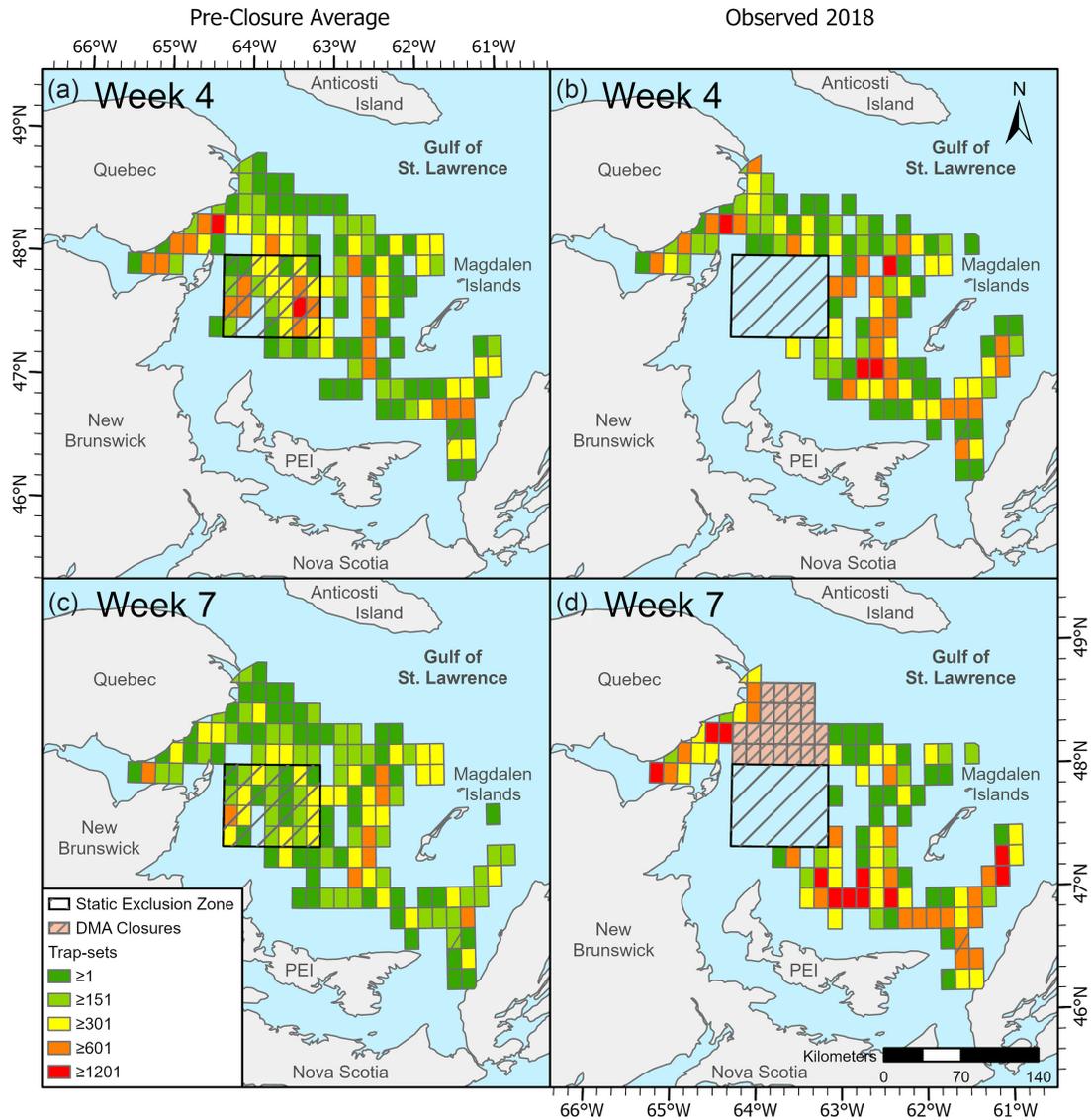


Figure 4. Fishing effort (trap-sets) distribution of the averaged pre-closure period without time-area closures (2015–2017; a, c) and the 2018 observed period (b, d) for Week 4 (a, b), and Week 7 (c, d) representing, respectively, the fully active fishery without and with dynamic closures. For comparability, the legend is the same for each map with dark green representing the lowest effort density at ≤ 150 trap-sets and red at the highest with ≥ 1201 trap-sets. Areas closed to fishing in 2018 are denoted by hatched lines, with the static closure denoted by a thick black outline and dynamic closures in light pink.

Discussion

This research demonstrates that time-area closures can alter the distribution and nature of a fishery in predictable (e.g. redistribution of effort from closed areas) and unpredictable (e.g. destination of displaced effort) ways. We presented evidence that the sGSL right whale time-area closures changed fishing effort density and distribution such that the entanglement threat increased in areas outside of the closed zones.

The size and location of the time-area closures strongly influenced the resulting distribution of the fishery as this affected the amount of fishing effort that was required to relocate and the amount of space available to which it could move (Murray *et al.*, 2000; Halpern *et al.*, 2004). The 2018 static closure was relatively large (approximately 6500 km²) and occurred in a highly produc-

tive area for the snow crab fishery. Closure size increased throughout the season due to the dynamic closures, further increasing the amount of fishing effort being displaced while reducing the number of viable cells available for fishing. Ultimately, approximately a third of the total fishing effort in the fishery was displaced due to the static and dynamic closures. Moreover, aside from a small reduction in the maximum trap limit in 2018 (returning to limits similar to years prior to 2017 when additional traps were added due to the large 2017 TAC), no additional effort controls were implemented. Fishing effort, therefore, continued at the same magnitude as comparable years prior to the establishment of the time-area closures (Hébert *et al.*, 2020) but in a smaller area, increasing the effort density outside the closed areas (a “squeezing” effect *sensu* Halpern *et al.*, 2004). As such, this increase in effort density resulted from less

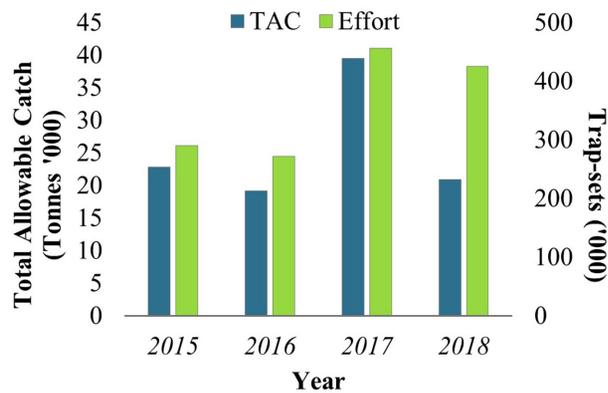


Figure 5. Annual snow crab TAC (tonnes; blue) and fishing effort (trap-sets; green) by year for crab fishing Area 12 in the Southern Gulf of St. Lawrence.

available and viable fishing areas, rather than because of an increase in the total number of traps. This was observed in both our model framework results and in the 2018 data (Figures 4 and 7).

Furthermore, overall fishing effort (i.e. trap-sets) in 2018 was equivalent to 2017 despite a smaller TAC and reduced trap limit (Hébert et al., 2020; Figure 5). This increased effort could only have resulted from traps being set and hauled more often. Two explanations for this are: (i) that fishers had difficulty catching their quota due to fishing in less productive grounds and thus increased the frequency of setting and hauling traps, and the total time spent fishing; and/or (ii) that fishers had to relocate more frequently (e.g. due to dynamic closures), and thus set and hauled traps more frequently.

While the increase in effort density outside the closed areas is important, it is equally as important to understand how the distribution of this effort changed. Following the implementation of the 2018 closures, effort shifted farther than anticipated, such that effort densities in the southeastern sGSL, nearer to PEI, were higher than previous years (Figures 4 and 7). Prior to the closures, the southeastern areas had lower effort, and therefore less entanglement threat when compared to the core fishing grounds where the static closure was delineated. However, with this physical shift in fishing effort, the southeastern sGSL contained an increased threat to whales when present. While time-area closures provide protection where whales are expected and observed to aggregate, the full extent of their habitat use in the sGSL is unknown and the risk outside the protected areas uncertain. Consequently, the shift we observed in this study provides an example of how displaced effort from time-area closures can result in unintended new areas of elevated threat (i.e. fishing effort density). Understanding these patterns of redistribution are important to ensure fishing effort continues to be effectively monitored and managed outside the closures to continue providing conservation benefits.

This observation of effort shifting away from the areas of the closures is contrary to other studies that suggests a “fencing” effect, or a disproportionate increase in fishing activity along the perimeter of a closure such that it encloses the closed area with fishing gear, following the implementation of time-area closures (Dinmore et al., 2003; Kellner et al., 2007; van der Lee et al., 2013). In this study, fewer boundary cells were fished in 2018 compared to the pre-closure period. However, this definition of a “fencing” effect often oversimplifies boundary threat, as the density of effort needs to be considered along with the spatial distribution of fishing along the perimeter of a closure. So, although fewer bound-

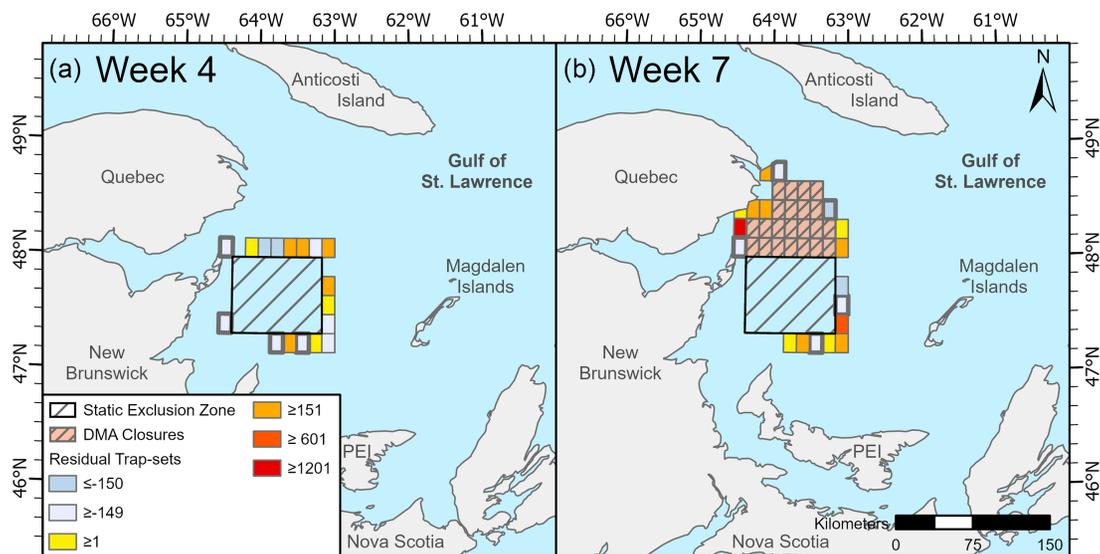


Figure 6. Fishing effort density in cells adjacent to 2018 time-area closures during Week 4 (a) and Week 7 (b), representing, respectively, the fully active fishery without and with dynamic closures. The value of each cell is the difference in trap-sets (i.e. residual) between the 2018 season and the averaged pre-closure period (2015–2017). The legend for both maps are the same where cool colours (negative values) represent a decrease in effort density in 2018 compared to the pre-closure period, and warm colours (positive values) represent an increase. Boundary cells with thick grey outlines were not fished in 2018. Areas closed to fishing in 2018 are denoted by hatched lines, with the static closure denoted by a thick black outline and dynamic closures in light pink.

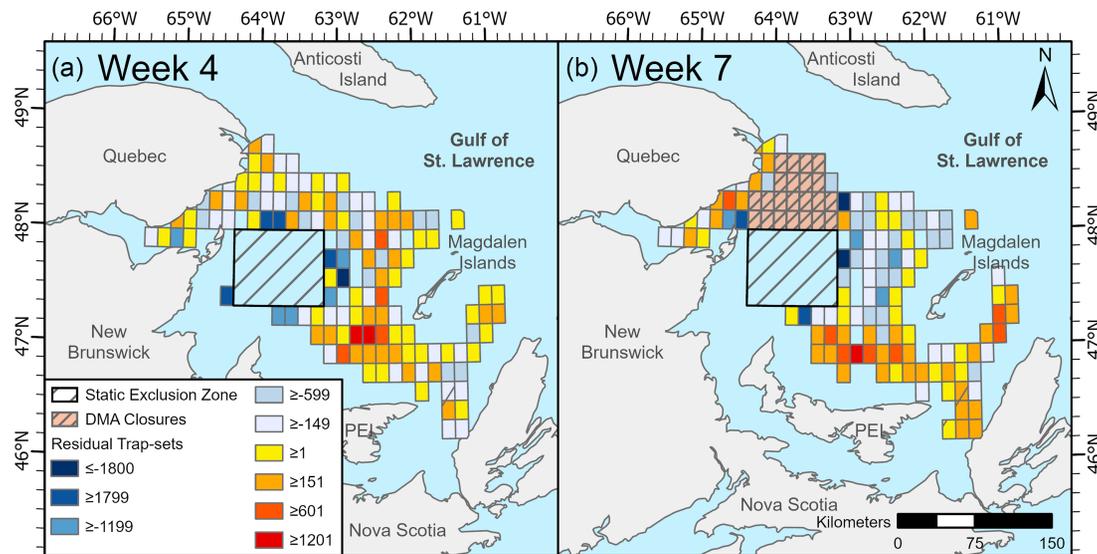


Figure 7. Difference in trap-sets (i.e. residual) between the observed period (i.e. 2018) and model predicted fishing effort distributions for Week 4 (a) and Week 7 (b), representing, respectively, the fully active fishery without and with dynamic closures. The legend is the same for both maps where cool colours (negative values) represent an overestimation of effort (i.e. observed effort was lower than predicted by the displacement model) and warm colours (positive values) represent an underestimation of effort. Areas closed to fishing in 2018 are denoted by hatched lines, with the static closure denoted by a thick black outline and dynamic closures in light pink.

ary cells along the closed areas were fished compared to the pre-closure period, the fishing effort density within those that were fished increased. The consequent effect on entanglement mitigation, and thus entanglement risk to whales is highly dependent on the movement patterns of the whales around the closures. For this reason while the perimeter was not further enclosed as a result of displaced fishing effort, as is often assumed, the increased effort density, and thus threat in the fished cells, cannot be ignored.

To predict the resulting distribution of the fishery, our model framework was based primarily on the assumption that fishers would attempt to maximize their returns (i.e. CPUE) relative to costs (i.e. distance to new fishing areas; Halpern *et al.*, 2004; Powers and Abeare, 2009). Yet, our results indicate this assumption was not broadly true, as fishers were instead observed moving farther away from the existing closures (Figure 4). Thus, fishers were not simply moving to maximize catch rates, and must consider other factors when deciding where to relocate. These factors may have included the expected probability of future dynamic closures, the potential for entanglements, or the proximity to suitable alternative ports (Murray and Ings, 2015). This differs notably from studies of time-area closures used to protect spawning stocks, where fishery catches along the perimeter can increase because of a “spill-over” effect (van der Lee *et al.*, 2013).

The reduction in cells available for fishing may also have influenced how fishers explored new fishing areas. Increased abandonment and reduced exploration of fishing areas during 2018 may have been due to fishers changing their fishing practices in response to the presence of time-area closures. For example, they may have focused on known fishing grounds, avoiding the risk of exploring new, but potentially less productive areas. Alternatively, fishers may have considered areas near the time-area closures to be unsuitable for crab fishing, and therefore moved farther away. For any of these explanations, it is clear that the behaviour of fishers strongly influences the resulting distribution of a fishery due to time-area

closures. Incorporating these additional motivations of fishers into future models would strengthen predictions of resulting distribution. It is also noteworthy, that because the right whale time-area closures for this fishery used a pre-existing protocol (i.e. soft-shell crab), fishers had prior experience with relatively small-scale closures, and thus an understanding of how to respond and adjust their fishing activity accordingly (Hébert *et al.*, 2020). Fisheries unfamiliar with the types of closures examined here, may respond differently than this fishery which is experienced with time-area closures.

While no right whale deaths occurred in Canadian waters in 2018, three right whales were entangled, two of which occurred in the sGSL (Pettis *et al.*, 2018). In 2019, which followed a similar time-area closure strategy to 2018, ten right whale mortalities were detected, with nine occurring in Canadian waters (Bourque *et al.*, 2020; Pettis *et al.*, 2020). Four of these nine whales were necropsied with one mortality directly attributed to entanglement in the sGSL. The tenth whale (whose carcass was found in US waters) was last sighted entangled in the sGSL before its death (Bourque *et al.*, 2020; Pettis *et al.*, 2020). An additional four live-entanglements also occurred in 2019, two of which were first sighted in the sGSL (Pettis *et al.*, 2020), indicating that entanglement threat is not completely abated in the sGSL through the use of time-area closures. Our results support these observations by demonstrating that there is an increased entanglement threat in the areas outside the time-area closures due to displaced fishing effort, and that this effort is shifting to previously low-threat areas which must continue to be accounted for (e.g. Murawski *et al.*, 2005; Hiddink *et al.*, 2006; Abbott and Haynie, 2012; Hoos *et al.*, 2019).

Time-area closures are an effective method of reducing entanglement risk for the areas where they are implemented (Murray *et al.*, 2000; Dawson and Slooten, 2005; Slooten, 2013), however, right whales are highly mobile and frequently move among regional habitats, even within seasons (Baumgartner and Mate,

2005; Brillant *et al.*, 2015; Bourque *et al.*, 2020). Fishing effort outside of time-area closures must, therefore, continue to be managed as an entanglement threat to whales. In addition to time-area closures, potential interventions to further assist fisheries in reducing entanglement threats include efforts to reduce the likelihood of entanglement (e.g. through the development of fishing gear that does not use persistent buoy lines, i.e. “ropeless” gear; Myers *et al.*, 2019), and to reduce the severity of injuries due to entanglements (e.g. through the use of reduced breaking strength rope; Knowlton *et al.*, 2016). Knowledge of how fishing distribution changes due to time-area closures, as shown in this study, can direct where these further interventions may be most effective. Lastly, the use of time-area closures must consider multiple conservation objectives so that the protection of one species does not increase risk to another. As time-area closures for cetacean protection are increasingly used globally, effects of multiple displaced fisheries must be assessed to consider changes in risk to all marine wildlife.

Supplementary data

Supplementary material is available at the ICES/JMS online version of the manuscript.

Data availability statement

The fisheries logbook data used for the effort calculations were provided by Fisheries and Oceans Canada (DFO) by permission. Data can be requested directly through the regional DFO departments. Ongoing fishery closures are publicly announced via DFO and can be accessed at <https://dfo-mpo.gc.ca/fisheries-peches/commercial-commercial/atl-arc/narw-bnan/index-eng.html>. All other data underlying this article will be shared on reasonable request to the corresponding author.

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