Screening for high-risk marine invaders in the Hudson Bay Region, Canadian Arctic: Compilation of background information, rationale, and references used to answer questions with the Canadian Marine Invasive Species Tool (CMIST)

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2023

SCREENING FOR HIGH-RISK MARINE INVADERS IN THE HUDSON BAY REGION, CANADIAN ARCTIC: COMPILATION OF BACKGROUND INFORMATION, RATIONALE, AND REFERENCES USED TO ANSWER QUESTIONS WITH THE CANADIAN MARINE INVASIVE SPECIES TOOL (CMIST)

by

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ABSTRACT

Goldsmit, J., Clark, H.A., McKindsey, C.W., Stewart, D.B. and Howland, K.L. 2023. Screening for high-risk marine invaders in the Hudson Bay Region, Canadian Arctic: Compilation of background information, rationale, and references used to answer questions with the Canadian Marine Invasive Species Tool (CMIST). Can. Data Rep. Fish. Aquat. Sci. 1373: vi + 344 p.

The present Data Report contains the documented information that was used during the horizon scanning to identify aquatic invasive species of potential concern to the Hudson Bay region, using the semi-quantitative Canadian Marine Invasive Screening Tool (CMIST). Complete scoring and background information is provided, together with the reference list used to assess a total of 31 species, including zoobenthos, phytobenthos and zooplankton. The present Data Report is complementary to the main publication: “Screening for high-risk marine invaders in the Hudson Bay Region, Canadian Arctic”, published in 2021 in Frontiers in Ecology and Evolution.

RÉSUMÉ

Goldsmit, J., Clark, H.A., McKindsey, C.W., Stewart, D.B. and Howland, K.L. 2023. Screening for high-risk marine invaders in the Hudson Bay Region, Canadian Arctic: Compilation of background information, rationale, and references used to answer questions with the Canadian Marine Invasive Species Tool (CMIST). Can. Data Rep. Fish. Aquat. Sci. 1373: vi + 344 p.

Le présent rapport de données contient les informations utilisées lors de l’analyse prospective des espèces aquatiques envahissantes potentiellement préoccupantes pour la région de la baie d'Hudson. Un outil semi-quantitatif, l’outil canadien d’évaluation préalable des risques (connue comme CMIST en anglais : Canadian Marine Invasive Screening Tool) a été utilisé. Les informations contextuelles ainsi que les références sont fournies pour l’établissement des cotes pour 31 espèces, y compris le zoobenthos, le phytobenthos et le zooplancton. Le présent rapport de données est complémentaire à la publication principale : "Screening for high-risk marine invaders in the Hudson Bay Region, Canadian Arctic", publiée en 2021 dans Frontiers in Ecology and Evolution.

INTRODUCTION

A horizon scanning exercise was conducted using the semi-quantitative Canadian Marine Invasive Screening Tool (CMIST) (Drolet et al., 2016) to identify aquatic invasive species (AIS) of potential concern to the Hudson Bay region. This region is considered to be at high risk for providing environmental conditions that are suitable for new species establishing (Ware et al., 2016; Goldsmit et al., 2018; 2020). Increased shipping activity and changes due to global warming increase the probability of new introductions (Melia et al., 2016; Essl et al., 2020). Documented information is used in this screening-level risk assessment tool to answer 17 questions related to the likelihood and impact of invasion. This data report contains the complete set of information collected to answer these questions together with the list of references used for a total of 31 species (zoobenthos, phytobenthos and zooplankton). Scores and analyses of CMIST results are included in the primary publication “Screening for high-risk marine invaders in the Hudson Bay Region, Canadian Arctic” (Goldsmit et al., 2021). This publication was part of the special issue ‘Invaders on the Horizon! Scanning the Future of Invasion Science and Management’ in the journal *Frontiers in Ecology and Evolution*.

MATERIAL AND METHODS

Description of the study area

The Hudson Bay Complex Large Marine Ecosystem (HBC LME) comprises Hudson Strait, Hudson Bay, Foxe Basin, James Bay and Ungava Bay (Figure 1) and it is one of eight marine ecoregions of the Canadian Arctic (Spalding et al., 2007). It receives a large volume of freshwater runoff, Arctic marine waters penetrate into the system, and the geomorphology along its coastal zone is dynamic (Stewart and Lockhart, 2005; and references therein). It is relatively shallow (150 m mean depth) (Prinsenberg, 1986) (Figure 1). Sea ice cover variations are dominated by inter-annual local atmospheric conditions (Hochheim and Barber, 2014; and references therein). A great proportion of ports in the Canadian Arctic are in the Hudson Bay Region (Chan et al., 2012, Goldsmit et al., 2019) (Figure 1).

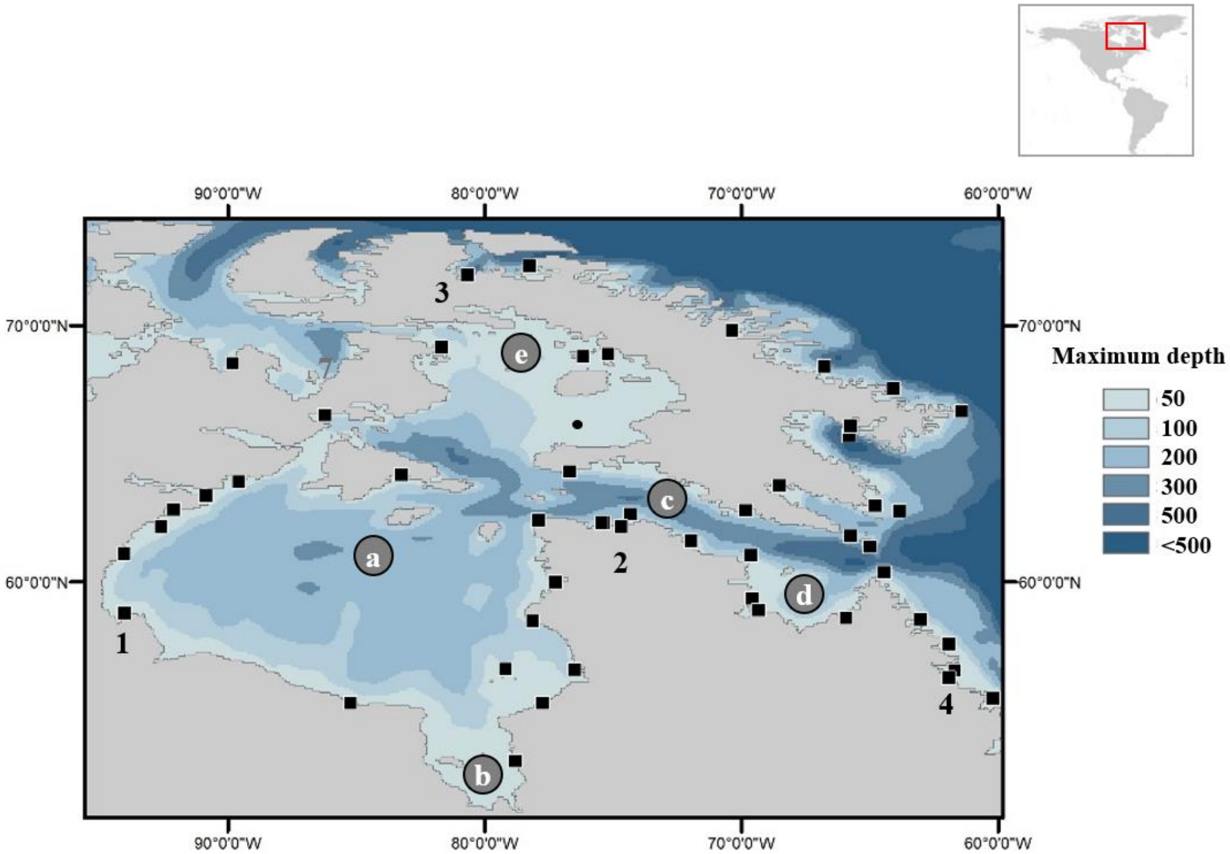

Figure 1: Risk assessment area. The Hudson Bay Complex Large Marine Ecosystem comprises (a) Hudson Bay; (b) James Bay; (c) Hudson Strait; (d) Ungava Bay; and (e) Foxe Basin. The black squares represent Canadian Arctic ports/locations that receive any type of shipment via commercial shipping (Chan et al. 2012; Goldsmit et al. 2019; DFO 2020). Numbers shown represent the main ports of the region where ballast water is discharged on a regular basis: (1) Churchill; (2) Deception Bay; (3) Milne Inlet; and (4) Voisey's Bay. Note that ports 3 and 4 are part of other ecoregions (Baffin Bay-Davis Strait and Northern Labrador, respectively), which are outside of the Hudson Bay Complex.

Description of the risk assessment tool

CMIST (Drolet et al., 2016) is a screening-level risk assessment (RA) tool that uses documented information and expert opinion to semi-quantitatively assess the risk of non-indigenous aquatic species. It consists of 17 questions related to the likelihood and impact of invasion and considers the level of certainty in the final score value. Assessors base their scores using a combination of expert knowledge and the best-available data (published articles, government reports, gray literature, and global invasive species websites) for each species assessed. At times, information on closely related species may be used to complete information not available for a given species. The complete list of CMIST questions are:

Likelihood of invasion

Q1: Is the species established in the assessment area?

Q2: How frequently and in what numbers is the species expected to arrive into the assessment area?

Q3: How much of the assessment area offers suitable habitat for the species?

Q4: How much of the assessment area offers suitable environmental conditions for the species to survive?

Q5: Are the species' reproductive requirements available in the assessment area?

Q6: To what extent could natural control agents slow the species' population growth in the assessment area?

Q7: What is the range of the species' potential natural dispersal in the assessment area?

Q8: What is the range of the species' potential dispersal in the assessment area from anthropogenic mechanisms?

Impact of invasion

Q9: What level of impact could the species have on population growth of other species in the assessment area?

Q10: What level of impact could the species have on communities in the assessment area?

Q11: What level of impact could the species have on habitat in the assessment area?

Q12: What level of impact could the species have on ecosystem function in the assessment area?

Q13: What level of impact could the species' associated diseases, parasites, or travelers have on other species in the assessment area?

Q14: What level of genetic impact could the species have on other species in the assessment area?

Q15: What level of impact could the species have on at-risk or depleted species in the assessment area?

Q16: What level of impact could the species have on aquaculture and commercially fished species in the assessment area?

Q17: Is the species known or generally considered to be invasive anywhere in the world?

Each question is scored from 1 (low) to 3 (high). Mean values of scores are calculated for Invasion and Impact and are then multiplied to yield a final risk score per species. Overall scores can range from 1 to 9 (lowest to highest respectively). A certainty score is also given to each question score, from 1 (low) to 3 (high). This step is done to account for confidence on the scoring, and it is related to the quality of information that was available when the assessment was performed.

A total of 31 species were ranked in Goldsmit et al. (2021). These included zoobenthos (amphipods, barnacles, crabs, molluscs, tunicates, and others), phytobenthos (macroalgae) and zooplankton (copepods and macrozooplankton) that may be transported by ship traffic, either in ballast water or as biofouling (Table 1).

Table 1: Species ranked using the CMIST tool.

Species	**Common name**	**Taxa**	**Taxonomic group**
Ecological group: Zoobenthos			
Caprella mutica	Japanese skeleton shrimp	Crustacea	Amphipod
Gammarus tigrinus	Tiger scud	Crustacea	Amphipod
Pontogammarus robustoides	Scud	Crustacea	Amphipod
Amphibalanus amphitrite	Striped barnacle	Crustacea	Barnacle
Amphibalanus eburneus	Ivory barnacle	Crustacea	Barnacle
Amphibalanus improvisus	Bay barnacle	Crustacea	Barnacle
Austrominius modestus	Australian barnacle	Crustacea	Barnacle
Carcinus maenas	Green crab	Crustacea	Crab
Chionoecetes opilio	Snow crab	Crustacea	Crab
Eriocheir sinensis	Chinese mitten crab	Crustacea	Crab
Paralithodes camtschaticus	Red king crab	Crustacea	Crab
Littorina littorea	Common periwinkle	Mollusca	Mollusc
Mya arenaria	Soft shell clam	Mollusca	Mollusc
Botrylloides violaceus	Violet Tunicate	Tunicata	Tunicate
Botryllus schlosseri	Golden star tunicate	Tunicata	Tunicate
Ciona intestinalis	Vase tunicate	Tunicata	Tunicate
Molgula manhattensis	Sea grape	Tunicata	Tunicate
Styela clava	Club tunicate	Tunicata	Tunicate
Cordylophora caspia	Freshwater hydroid	Cnidaria	Other
Marenzelleria viridis	Red-gilled mudworm	Polychaeta	Other
Membranipora membranacea	Coffin box bryozoan	Bryozoa	Other
Ecological group: Phytobenthos			
Codium fragile spp. *fragile*	Dead man's fingers	Chlorophyta	Macroalga
Dumontia contorta	Dumont's tubular weed	Rhodophyta	Macroalga
Sargassum muticum	Japanese wireweed	Phaeophyceae	Macroalga
Undaria pinnatifida	Wakame	Phaeophyceae	Macroalga
Ecological group: Zooplankton			
Acartia (Acanthacartia) tonsa	No common name found	Copepoda	Copepod
Centropages typicus	No common name found	Copepoda	Copepod
Eurytemora affinis affinis	No common name found	Copepoda	Copepod
Eurytemora carolleeae	No common name found	Copepoda	Copepod
Aurelia limbata	Brown banded moon jelly	Cnidaria	Macrozooplankton
Mnemiopsis leidyi	Warty comb jelly	Ctenophora	Macrozooplankton

The results section of the present Data Report contains all the information used in Goldsmit et al. (2021) to score each CMIST question for all species listed above. No new information has been added since the publication of this paper.

DISCLAIMER: information used to answer the questions was often copied directly from the cited documents. This was done to maintain the integrity of the information provided. The main sources cited include primary and secondary publications (reports, etc.) as well as various on-line databases, including global invasive species lists (National Exotic Marine and Estuarine Species Information System NEMESIS: https://invasions.si.edu/nemesis/; Invasive Species Compendium: www.cabi.org/isc; the European Network on Invasive Alien Species NOBANIS: www.nobanis.org/; the Global Invasive Species Database GISD: www.issg.org/database; the Global Biodiversity Information Facility GBIF: https://www.gbif.org/; AquaMaps: https://www.aquamaps.org/; the World Registry of Marine Species WORMS: http://www.marinespecies.org/index.php, etc.)

RESULTS

ZOOBENTHOS - AMPHIPODS

Caprella mutica Schurin, 1935

Phylum: Arthropoda
Class: Malacostraca
Order: Amphipoda
Family: Caprellidae

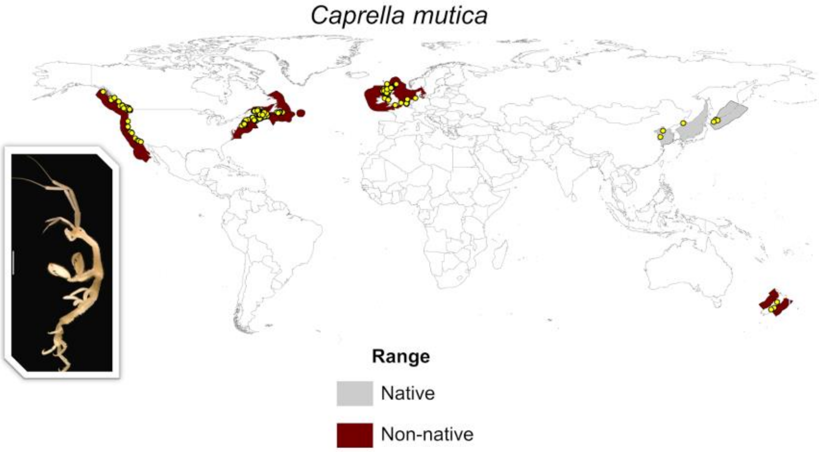

Figure 2: Ecoregions where Caprella mutica is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Caprella mutica occurrence points were obtained from OBIS (https://obis.org/) and NEMESIS (http://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-77). Picture of C. mutica modified from https://invasions.si.edu/nemesis/species_summary/-77

CMIST scores for *C. mutica:*

Mean adjusted Likelihood of Invasion: 2.22

Mean adjusted Impact of Invasion: 1.93

Mean adjusted Overall CMIST score: 4.29

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggest *Caprella mutica* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Goldsmit, 2016) but *Caprella* sp. have been reported from Deception Bay (Goldsmit et al., 2014) and the Churchill estuary (Baker et al., 1994). It is common along the British Columbian (BC) coast and on domestically operated commercial vessels (Frey et al., 2009). It is established in the Maritimes and Gulf of St. Lawrence and has extended its range there (Prince Edward Island (PEI) since 1998; Quebec (QC) 2003; Nova Scotia (NS) 2012) (Turcotte and Sainte-Marie, 2009, Vercaemer et al., 2012).

Q2- Rate of introduction (Score = 3, Certainty = 2)

Score Rationale: Reliable information suggests *Caprella mutica* could arrive frequently in high numbers in the RA area. The species has been found on arriving vessels and is present in a number of connected port ecoregions.

Background Information: Found in biofouling assemblages in ships traveling to Canadian Arctic ports (in all ships sampled it was found to be present before going to the Arctic, n=8) (voyage 1= 10 individuals; voyage 2= 3 individuals; voyage 3= 41 individuals; voyage 4= 30 individuals; voyage 5= 71 individuals; voyage 6= 4 individuals; voyage 7= 7 individuals; voyage 8= 277 individuals) (Chan et al., 2015). It is abundant and widespread in ecoregions of connected ports on both coasts of North America and Europe and in adjacent ecoregions (Newfoundland) and could be transported on commercial vessels or on fishing boats/gear. Caprellid survival in ballast water has been demonstrated (Carlton, 1985, Cook et al., 2007, Turcotte and Sainte-Marie, 2009).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Caprella mutica*. There is reliable information about seabed morphological characteristics in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: From Turcotte and Sainte-Marie (2009) and references therein: “Caprellids in general are associated with littoral and infralittoral habitats, colonizing structures such as algae, hydrozoans and bryozoans. In their native range in the Sea of Japan, it occurs naturally in the infralittoral zone (0.7-13 m deep) in eelgrass or algal beds and on drifting macroalgae. In its invaded range, it is mainly found on artificial structures like nets, ropes and cages used in aquaculture, docks, oil rigs and navigation buoys.” Buschbaum and Gutow (2005) hypothesized that this association with artificial structures could be due to the fact that these substrates represent a transitional habitat where *C. mutica* may acclimatize for a few years before spreading to natural habitats in its new environment.

From NEMESIS: “In the North Sea, off Netherlands and Belgium, *C. mutica* was dominant on intertidal and floating artificial substrates, in waters with high densities of suspended particulate matter, and less than 17 m depth (Coolen et al., 2016). *C. mutica* appears to have sufficient tolerance and flexibility of habitat, feeding, and life history to colonize much of the world's temperate waters (Ashton, 2006, Boos et al., 2011).” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-77)

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Caprella mutica*. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From NEMESIS: “It tolerates wide ranges of temperature, 2 - 25^{0}C, and salinities as low as 11 PSU in the field in the northern Sea of Japan (Shevchenko et al., 2004).” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-77).

From Boos et al. (2011): “Based on field distribution minimum temperature is -2^{0}C (Ashton et al., 2007). Lethargic at low temperatures (2^{0}C) after 48 hours in the laboratory, longer periods at low temperatures (4^{0}C) surviving up to 5 months in the lab (Ashton et al., 2007, Boos, 2009). Annual temperatures in the species' native area can range between -1.8 and 25^{0}C (Shevchenko et al., 2004). Species will be excluded from brackish water environments such as the heads of sea-lochs or estuaries and semi-enclosed areas such as marinas with freshwater input.”

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Caprella mutica* in the RA area. There is a reasonable amount of experimental data, but results are variable, depending on the population tested. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >=13 °C resulted in selecting a small region in James Bay. It has been seen that *C. mutica* can reproduce at 0 and 2 °C, and the minimum temperature tested was 4 °C. When regions of the assessment area were calculated as >=4 °C, it showed a moderate extension of suitable habitat. This temperature range overlaps with salinity conditions, so it could be scored as "sometimes" having suitable reproductive conditions available.

Background Information: In the lab, females showed high productivity at 13-14^{0}C. In their native range, egg incubation lasted 30-40 days at temperatures varying between 13^{0}C and 17^{0}C and 60 days at temperatures varying between 0 and 2^{0}C. Hosono (2009) experimentally demonstrated that *C. mutica* reproduces at temperatures ranging from 5 to 20°C, but found the mean oviposition–juvenile emergence period was longer at lower temperatures. They found eggs developed normally at 15 and 20°C but showed decreased viability at 5 or 10°C. Hosono (2011) experimentally showed that juvenile *C. mutica* died by the third instar at temperatures of 5°C, while animals reared at 10, 15 and 20°C lived to higher instars and reached maturity within their lifetime. However, Ashton et al. (2007) found survival of hatchlings for up to 4 months at the tested minimum reproductive temperature of 4°C and minimum salinity of 15 PSU, and suggested that winter hatchlings could likely survive with spring warming. Ashton et al. (2007), quoting Fedotov (1991), states that “the reproductive period stretched from March to July (average water temperature of between 0 and 17.4 °C, respectively), reaching a peak in March when 72.7% of the total females carried eggs.” There is no planktonic larval dispersal phase, embryonic and larval development occurs inside the egg, juvenile caprellids remain in the marsupial chamber for a period between a few hours and 3-4 weeks after emerging. Direct development and maternal care likely provide a very high survival rate for the progeny. Although the occurrence of maternal care has not yet been demonstrated in *C. mutica*, it has been seen in other species of the genus.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Caprella mutica*. There are likely closely related species in the RA area that could act as predators.

Background Information: Known natural predators of caprellids are primarily fish, followed by crabs, nudibranchs, starfish and hydrozoans (Turcotte and Sainte-Marie (2009) and references therein). There is no known disease specific to *C. mutica,* although a parasitic copepod has been found within the brood pouch of certain caprellids, which mimics the morphology of its eggs (Huys pers. comm. 2009, from Boos et al. (2011).

From Turcotte and Sainte-Marie (2009): “The known natural predators of caprellids are primarily fish (e.g. Caine, 1989, 1991, Page et al., 2007, O’Gorman et al., 2008), and to a lesser extent, invertebrates such as crabs (Dubiaski-Silva and Masunari, 2008), nudibranchs (Caine, 1980), starfish (Lauerman, 1998) and possibly hydrozoans (Genzano, 2005). Caprellids can represent a significant link in the food chain between unicellular algae and some predatory fish (Caine, 1989, Holbrook and Schmitt, 1992, O’Gorman et al., 2008). In fact, it was recently proposed that caprellids could be used as fish feed in aquaculture operations (Woods, 2009).

Fish are visual predators of caprellids and select their prey based on their behaviour (e.g. movement and degree of exposure) and size (Caine, 1989). The passage of the migratory fish *Cymatogaster aggregata* was linked to a marked decrease of *C. laeviuscula* populations in Padilla Bay in Washington State, on the American west coast (Caine, 1991). Caprellids were not the preferred prey of *C. aggregata*, but rather a temporary diet option for a few weeks during a critical period of the fish life cycle (Caine, 1991). However, other fish species or subpopulations of a fish species can be specialized caprellid predators (Holbrook and Schmitt, 1992). It is possible that male caprellids are more vulnerable to predation by fish because of their larger size (Ashton, 2006)."

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Caprella mutica* has the capacity for a moderate range of natural dispersal in the RA area. Background information suggests this mainly occurs through passive drifting, but there is some uncertainty as to the magnitude of dispersal through this mechanism.

Background Information: From Turcotte and Sainte-Marie (2009): “*C. mutica* does not have a larval phase that can be dispersed by currents, but some other species can move actively by crawling or swimming and passively by drifting on floating algae (Caine, 1980). Crawling and swimming are probably not that effective for dispersal. It is more likely that the main dispersal mode of *C. mutica* on a medium spatial scale is passive and that it occurs by drifting (rafting) on natural floating objects, such as macro-algae and tree trunks, or on man-made waste (Ashton, 2006, Cook et al., 2007, Astudillo et al., 2009). These transportation vectors have the advantage of offering a structurally complex habitat protecting the caprellids from desiccation and predation and allowing them to continue their regular activities including reproduction (Highsmith, 1985, Thiel and Gutow, 2004). Sano et al. (2003) observed *C. mutica* individuals in drifting algae communities in the species’ natural habitat in Japan. *C. mutica* was also observed on *S. muticum* colonies floating on the surface near Helgoland Island in Germany and the drifting algae is considered the introduction and spreading mechanism of *C. mutica* in this area (Buschbaum and Gutow, 2005). In Scotland, 29.6% (7/26) of the drifting algae samples collected within a radius of 25 m from a vessel over a distance of 40 km contained *C. mutica* at a density reaching as many as 71 individuals per sample (Ashton, 2006). Passive transportation of *C. mutica* by drifting algae or debris could explain the rapid propagation of the species on the North American west coast and in Europe (e.g. Cook et al. (2007))."

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Caprella mutica* has the capacity for a wide range of anthropogenic dispersal in the RA area. The species is directly connected to areas where it is present and it has been shown to be present on hulls of ships going directly to the assessment region and may also be transported in ballast.

Background Information: From Turcotte and Sainte-Marie (2009) and references therein: “The transoceanic spreading mechanisms are likely through the transfer of Japanese oysters, ballast water and to a lesser degree, the hulls of vessels. All reported introductions have occurred in areas with human activities: marinas, ports or aquaculture sites. In its native range it can be found in association with the brown algae *Sargassum muticum*, which has been also related to the oysters exportation. Nevertheless, commercial navigation is thought to be the primary vector of introduction in Europe.

Caprellid survival in ballast water has been demonstrated (Carlton, 1985). Secondary dispersal can be achieved by fouling smaller domestic or pleasure vessels, allowing the species to travel against currents and colonize areas upstream."

C. mutica has been found in biofouling assemblages in ships traveling to Canadian Arctic ports (Chan et al., 2015). Hull fouling and ballast movement by domestic coastal shipping could spread *C. mutica* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012, Chan et al., 2015). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total amount) in the region, suggesting that this is a plausible vector for within-region spread. With the exception of ship-related movements, the importance of other types of movements (e.g., smaller pleasure vessels) in HBC is not well known.

Q9- Impact on population (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Caprella mutica* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. There is some information about ecological effects on populations and communities, and thus ecosystem functioning, but these appear to be moderate and information is sometimes conflicting.

Background Information: From Turcotte and Sainte-Marie (2009) and references therein: "Despite the introduction of *C. mutica* in several countries, few direct or indirect impacts have been reported. *C. mutica* population dynamics suggest that its impact is most likely to be felt during the summer because of its high abundance (Ashton, 2006). Mussel farmers from Scotland and Quebec have reported a decrease in *Mytilus* spp. spat collection and growth since the appearance of *C. mutica* (Ashton, 2006). A causal relationship could not be confirmed in Scotland (Ashton, 2006). However, field and laboratory work conducted in Quebec confirmed there was a negative effect of *C. mutica* on mussel spat. The underlying mechanisms were documented and mitigation measures were recommended.

The potential effects on the food chain structure and on the upper trophic levels represent a major concern regarding the introduction of an exotic species in an ecosystem (Grosholz, 2002). The introduction of *C. mutica* in California led to a noticeable change in the diet of the fish *Oxylebius pictus* on artificial reefs where this caprellid was abundant compared to the neighbouring natural reefs where it was not (Page et al., 2007). The transfer of predation effort from a native species towards a more abundant and available invasive species can have significant consequences on the predator which may be advantaged or disadvantaged depending on the circumstances (Page et al., 2007). One of the advantages could be a high abundance of food, whereas one of the disadvantages could be that the invasive species has a lower nutritional value than the native prey. The fatty acids in *C. mutica*, described by Kawashima et al. (1999), appear however to make it a quality food for fish. It should be noted that Takeuchi et al. (2001) noticed a sizable bioaccumulation of the contaminant butyltin in *Caprella* spp. and, by this way, the genus could be an excellent bio-indicator of pollution.

The introduction and development of *C. mutica* populations could however have other consequences on the native food web. *C. mutica* can no doubt represent an added food source for certain micro- or macro-predators and could improve their condition, but its abundance could also decrease the diversity and abundance of native invertebrate species, at least on artificial structures (Page et al., 2007). In its exotic distribution range, *C. mutica* does not appear to colonize natural substrates. It is therefore difficult to establish the potential effects of *C. mutica* on natural ecosystems. However, a laboratory study demonstrated the aggressive nature of *C. mutica* towards a caprellid species that is native to eastern Canada, *C. linearis*, and its ability to exclude and kill it. At high densities, *C. mutica* could entirely exclude *C. linearis* from habitats, whether they had shelters or not (Shucksmith et al., 2009)."

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Caprella mutica* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There is some information of ecological effects on populations and communities, and thus ecosystem functioning, but these appear to be moderate and information is sometimes conflicting.

Background Information: From NEMESIS: "Studies of its economic and ecological impacts are limited, but observations indicate that *C. mutica* can affect aquaculture operations, displace native caprellids, and affect the feeding of native fishes (Ashton, 2006, Page et al., 2007, Shucksmith et al., 2009, Turcotte and Sainte-Marie, 2009)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-77)

From Turcotte and Sainte-Marie (2009) and references therein: "Despite the introduction of *C. mutica* in several countries, few direct or indirect impacts have been reported. *C. mutica* population dynamics suggest that its impact is most likely to be felt during the summer because of its high abundance (Ashton, 2006). Mussel farmers from Scotland and Quebec have reported a decrease in *Mytilus* spp. spat collection and growth since the appearance of *C. mutica* (Ashton, 2006). A causal relationship could not be confirmed in Scotland (Ashton, 2006). However, field and laboratory work conducted in Quebec confirmed there was a negative effect of *C. mutica* on mussel spat. The underlying mechanisms were documented and mitigation measures were recommended.

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Q11- Impact on habitats (Score= 1, Certainty= 2)

Score Rationale: Some reliable information suggests *Caprella mutica* would have low or no impact on habitat in the RA area. Although there is a reasonable amount of published information on other impacts due to this species, there is no clear evidence of impacts on habitats.

Background Information: From Turcotte and Sainte-Marie (2009) and references therein: "The introduction and development of *C. mutica* populations could however have other consequences on the native food web. *C. mutica* can no doubt represent an added food source for certain micro- or macro-predators and could improve their condition, but its abundance could also decrease the diversity and abundance of native invertebrate species, at least on artificial structures (Page et al., 2007). In its exotic distribution range, *C. mutica* does not appear to colonize natural substrates. It is therefore difficult to establish the potential effects of *C. mutica* on natural ecosystems. However, a laboratory study demonstrated the aggressive nature of *C. mutica* towards a caprellid species that is native to eastern Canada, *C. linearis*, and its ability to exclude and kill it. At high densities, *C. mutica* could entirely exclude *C. linearis* from habitats, whether they had shelters or not (Shucksmith et al., 2009)."

Q12- Impact on ecosystem function (Score= 2, Certainty= 2)

Score Rationale: Some reliable information suggests *Caprella mutica* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. There is some information on ecological effects on populations and communities, and thus ecosystem functioning.

Background Information: From Turcotte and Sainte-Marie (2009) and references therein: "Can have consequences on the native food web. *C. mutica* can represent an added food source for some predators and could improve their conditions, but its abundance could decrease the diversity and abundance of native invertebrate species at least on artificial structures (Page et al., 2007). It is difficult to establish the potential effects on natural ecosystems.

"Despite the introduction of *C. mutica* in several countries, few direct or indirect impacts have been reported. *C. mutica* population dynamics suggest that its impact is most likely to be felt during the summer because of its high abundance (Ashton, 2006). Mussel farmers from Scotland and Quebec have reported a decrease in *Mytilus* spp. spat collection and growth since the appearance of *C. mutica* (Ashton, 2006). A causal relationship could not be confirmed in Scotland (Ashton, 2006). However, field and laboratory work conducted in Quebec confirmed there was a negative effect of *C. mutica* on mussel spat. The underlying mechanisms were documented and mitigation measures were recommended.

The potential effects on the food chain structure and on the upper trophic levels represent a major concern regarding the introduction of an exotic species in an ecosystem (Grosholz, 2002). The introduction of *C. mutica* in California led to a noticeable change in the diet of the fish *Oxylebius pictus* on artificial reefs where this caprellid was abundant compared to the neighbouring natural reefs where it was not (Page et al., 2007). The transfer of predation effort from a native species towards a more abundant and available invasive species can have significant consequences on the predator which may be advantaged or disadvantaged depending on the circumstances (Page et al., 2007). One of the advantages could be a high abundance of food, whereas one of the disadvantages could be that the invasive species has a lower nutritional value than the native prey. The fatty acids in *C. mutica*, described by Kawashima et al. (1999), appear however to make it a quality food for fish. It should be noted that Takeuchi et al. (2001) noticed a sizable bioaccumulation of the contaminant butyltin in *Caprella* spp. and by this way, the genus could be an excellent bio-indicator of pollution.

The introduction and development of *C. mutica* populations could however have other consequences on the native food web. *C. mutica* can no doubt represent an added food source for certain micro- or macro-predators and could improve their condition, but its abundance could also decrease the diversity and abundance of native invertebrate species, at least on artificial structures (Page et al., 2007). In its exotic distribution range, *C. mutica* does not appear to colonize natural substrates. It is therefore difficult to establish the potential effects of *C. mutica* on natural ecosystems. However, a laboratory study demonstrated the aggressive nature of *C. mutica* towards a caprellid species that is native to eastern Canada, *C. linearis*, and its ability to exclude and kill it. At high densities, *C. mutica* could entirely exclude *C. linearis* from habitats, whether they had shelters or not (Shucksmith et al., 2009)."

Q13- Associated diseases, parasites, and travellers (Score= 1, Certainty= 1)

Score Rationale: Little to no reliable information is available to suggest *Caprella mutica* would have low or no impact associated with diseases, parasites or travelers in the RA area. There is no published evidence of impacts from disease/parasites, but this has not been evaluated to our knowledge, so low certainty.

Background Information: From (Boos et al., 2011) and references therein: "There is no known disease selective for *C. mutica*, although a parasitic copepod has been found within the brood pouch of certain caprellids, which mimics the morphology of the eggs. Further investigations on the impact that this parasite may have on the success of *C. mutica* in colonizing new habitats requires further investigation."

From Bateman and Stentiford (2017): "The natural host range of Hepatopancreatic Parvovirus (HPV) appears to include wild and cultured shrimp from across the Indo-Pacific region, Africa, Middle East, and the Americas (Lightner, 1993) and Madagascar (Tang et al., 2008). To date, over 100 species of arthropods have been reported as hosts or carriers of White Spot Syndrome Virus (WSSV) either from farm, wild or, experimental studies (Sánchez-Paz, 2010). Although the rate of detection has been steady since the discovery of the first crustacean virus in the 1960s, the majority of studies have focused on commercially or economically important hosts, such as penaeids, with very little assessment of infection in the wide diversity of lesser studied taxa. Given the diversity and abundance of crustaceans, particularly in aquatic environments, it can be assumed that a large number of novel viral taxa, in a wide array of known and, as yet unknown families, will remain to be discovered (Shi et al., 2016, Simmonds et al., 2017)."

Q14- Genetic impact (Score= 2, Certainty= 1)

Score Rationale: Little to no reliable information is available to suggest *Caprella mutica* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native caprellids of the same genus in the RA area, but no evidence of hybridization found in the literature.

Background Information: *C. septentrionalis* has been reported widely in the HB LME (Stewart and Lockhart, 2005). Native *Caprella* spp. found in HBC (https://www.gbif.org). There is no information found on hybridization of *Caprella* species.

Q15- Impact on at-risk species (Score= 2, Certainty= 1)

Score Rationale: Little to no reliable information is available to suggest *Caprella mutica* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be impacted given the demonstrated moderate effects on ecosystems elsewhere, however interactions and potential effects in the RA are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through food web and ecosystem impacts - these include: northern wolffish (*Anarhichas denticulatus*), spotted wolffish (*Anarhichas minor*), and the thorny skate (*Amblyraja radiata*). Walrus populations of risk/special concern may also be impacted as they feed on benthic invertebrates so may be impacted through food web changes in benthic habitats.

Q16- Impact on fisheries (Score= 2, Certainty= 2)

Score Rationale: Some reliable information suggests *Caprella mutica* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. Mussel farmers in a number of regions suggest the species may reduce spatfall. Impacts to fisheries/aquaculture species would likely not be great as aquaculture is not practiced in the area.

Background Information: Mussel farmers from Scotland and Quebec have reported a decrease in *Mytilus* spp spat collection and growth (Turcotte and Sainte-Marie, 2009). Natural populations of blue mussel (*Mytilus edulis*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, and occasionally for subsistence by Hudson Bay residents (Stewart and Lockhart, 2005). Damage to mussel populations in the Hudson Bay LME could also affect Hudson Bay eider (*Somateria mollissima sedentaria*) that overwinter in polynyas in the Belcher Islands, where they eat mussels and are hunted for subsistence.

Q17- Past invasion history (Score= 3, Certainty= 3)

Score Rationale: Considerable reliable information suggests *Caprella mutica* is invasive elsewhere in the world. This is a well-known invasive species with a large temperate invaded range and much has been published on various parts of its biology, life history, and invasion history.

Background Information: From NEMESIS: “It is introduced to the East (Connecticut to Quebec) and West coasts (California to Alaska) of North America, Europe (from France to Norway and Germany), and New Zealand. It is native to the Northwest Pacific and was first described in 1939 from Peter the Great Bay, Russia.” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-77).

Gammarus tigrinus Sexton, 1939

Phylum: Arthropoda

Class: Malacostraca

Order: Amphipoda

Family: Gammaridae

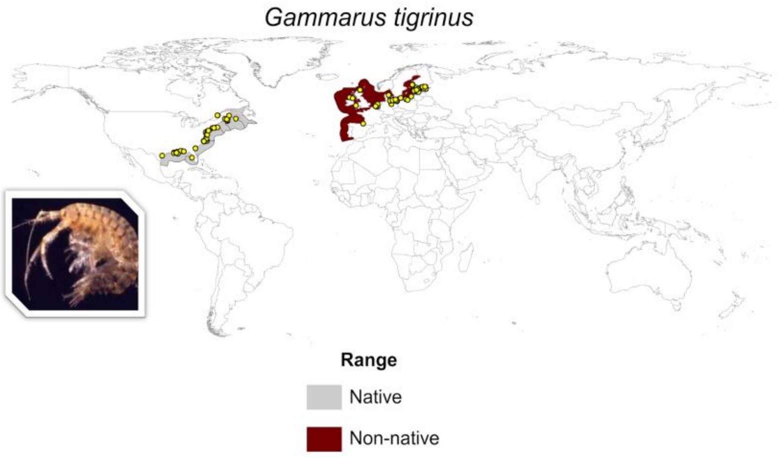

Figure 3: Ecoregions where Gammarus tigrinus is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Gammarus tigrinus occurrence points were obtained from OBIS (https://obis.org/). Picture of G. tigrinus modified from https://invasions.si.edu/nemesis/species_summary/93781

CMIST scores for *G. tigrinus:*

Mean adjusted Likelihood of Invasion: 2.20

Mean adjusted Impact of Invasion: 2.28

Mean adjusted Overall CMIST score: 5.03

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Gammarus tigrinus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There are no reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Goldsmit, 2016).

Q2- Rate of introduction **(Score = 3, Certainty = 2)**

Score Rationale: Some reliable information suggests *Gammarus tigrinus* could arrive frequently in high numbers in the RA area. The species has been found on arriving vessels and is present in a number of connected port ecoregions.

Background Information: Ships visiting Churchill can carry live *Gammarus* spp. on their hulls (Chan et al., 2015). Exposure occurs during the open water shipping season. *Gammarus tigrinus* is present in ports that are directly connected by shipping to Churchill and Deception Bay and could potentially be entrained and transported by these vessels in ballast water (Chan et al., 2012). It is also distributed from the Great Lakes to St. Lawrence River in Quebec and south along the eastern seaboard, so domestic or international vessels could be a very likely vector of introduction (https://www.gbif.org).

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Gammarus tigrinus*. Reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: From Ba et al. (2010): "It is dominant in intertidal and subtidal benthic habitats including reeds, algae, hard or soft substratum and sand (Bousfield, 1958, Bousfield, 1973, van Maren, 1978). In its native range, this euryhaline species lives in both fresh and brackish water (Bousfield, 1973), whereas in coastal waters it is restricted to shallow lagoons, bays, and estuaries."

Gammarus tigrinus shows low habitat selectivity (Bousfield, 1973). In invaded ranges *G. tigrinus* is more abundant in sheltered to moderately exposed coasts, with low salinity, and moderate-high eutrophication level. Wave-exposed shores, offshore reefs and similar habitats seem not favourable for the invasion of *G. tigrinus* (Packalén et al., 2008). The species has a broad ecological niche, but preference for sheltered shallow water areas with soft sediments (Reisalu et al., 2016). *Gammarus tigrinus* is attracted to sheltered habitats, including debris, algae, submerged and emergent vascular plants, hydroids (*Cordylophora caspia*), pilings, pebbles, and freshwater seeps and pools on rocky shores (Hynes, 1955, Bousfield, 1973, Kotta et al., 2011). *Gammarus* spp. are omnivores, feeding on algae, aquatic plants, and invertebrates, including juvenile amphipods of their own and other species (Jänes et al., 2015).

Q4- Survival (suitable environment) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Gammarus tigrinus*. A moderate amount of the RA area has the reported salinity tolerance values of the species (<= 18 and <= 25 PSU). In this case, salinity is the variable that determines most of the environmental suitability, rather than temperature, as it can tolerate cold temperatures throughout the RA area (<0° C). Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From NOBANIS: "Suitable salinity for *Gammarus tigrinus* are oligohaline (0.5-5(7.5) ppt) and mesohaline (5(7.5)-18 ppt) brackish waters" (https://www.nobanis.org/globalassets/speciesinfo/c/caprella-mutica/caprella-mutica2.pdf).

From Ba et al. (2010): "The species has a high tolerance toward salinity changes (0–25 PSU) and human degradation of the environment (Grabowski et al., 2007a, Grabowski et al., 2007b)."

From Kotta et al. (2013): "*Gammarus tigrinus* can withstand low temperatures between 1 or 2°C (Wijnhoven et al., 2003), but they are mainly limited by salinity. Strong tolerance to adverse environmental conditions (Sareyka et al., 2011). They are experimentally shown to tolerate better oxygen depletion, eutrophication and heat stress compared to native gammarids (Grabowski et al., 2007b, Sareyka et al., 2011)."

From Van der Velde et al. (2000): "Under laboratory conditions specimens of *G. tigrinus* survived in saline media with a conductivity range of 1000 - 20000 uS/cm at temperatures of 15-25 °C (Savage, 1982). It tolerates salinities of 1-16% (Dorgelo, 1974)."

From NEMESIS: "They are distributed as far north as the Great Lakes and marine portions of St. Lawrence River in Quebec where waters reach sub-zero temperatures (https://www.gbif.org). They tolerate a wide temperature range, from near freezing to above 31°C (Bousfield, 1973, Wijnhoven et al., 2003, Lenz et al., 2011). It has extensively colonized non-tidal waters in Europe, the Great Lakes, and the upper Mississippi River (Bousfield, 1958, Grigorovich et al., 2008). This amphipod is most abundant at salinities from 1-25 PSU, but can tolerate salinities up to 35 PSU (Dorgelo, 1975). However, different genetic lineages appear to vary in their salinity tolerance and preference, and ability to invade fresh or brackish water (LeCroy, 2000, Kelly et al., 2006a, Kelly et al., 2006b).” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=93781)

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are sometimes available for *Gammarus tigrinus* in the RA area. The species requirements are well demonstrated and there are reliable environmental data layers for the RA to show a match. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperatures >=5 °C (limit below which reproduction stops) and >=16 °C (limit for reaching sexual maturity) for reproduction in oligohaline regions, which resulted in some points all along the RA area that can reach these conditions during summer. The same calculation was done at >=10 °C and it still showed a moderate amount of suitable habitat in the RA area.

Background Information: Reproduction stops at or below 5°C (Pinkster et al., 1977). Reproductive cycles proceed more slowly when water temperature is lower (e.g.; English and Canadian populations reproduce slower than in the Netherlands) (Pinkster et al., 1977). Egg incubation period and growth rate are dependent upon temperature, reaching sexual maturity at 16 - 20°C in the Netherlands (Chambers, 1977). There is great reproductive capacity in oligohaline waters (Pinkster, 1975, Pinkster et al., 1977). The species is able to form populations in a wide range of inland and coastal ecosystems within temperate regions. It has a large brood size and the species mature at a smaller size and earlier than native gammarids in the Northern Baltic Sea (Sareyka et al., 2011). Nevertheless, there is variation from area to area with this species, so generalization must be considered with care. Variations may be caused by factors like temperature, food supply, water chemistry, intensity of predation, etc. (Chambers, 1977). At summer temperature, egg incubation period is about 10 days (Chambers, 1977). Rapid growth rate, early onset of sexual maturity and high fecundity enabled *G. tigrinus* to replace other gammarid species in Norwegian lakes (Chambers, 1977) and theoretically, 15-16 generations are possible during one year (Pinkster, 1975).

Q6- Establishment (natural control agents) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests natural control agents are unlikely to affect population growth in the RA area for *Gammarus tigrinus*. Available information shows this species has relatively low vulnerability to heavy predation and tends to be competitively superior in invaded regions, especially in turbid waters (that may limit visual predators) which are present throughout much of the RA area. There is some reliable information, but mainly generalities, and no species-specific information on predation related to native predators from the RA area; as well no information on whether or not the known parasites/diseases would be present in the RA area or transported with the host species in vectors.

Background Information: From Ba et al. (2010): “*Gammarus tigrinus* is well adapted to withstand predation (Grabowski et al., 2007a, Grabowski et al., 2007b). At present, it is difficult to foresee natural pressures which may limit the spread or establishment of this alien species.”

From CABI: "Fishes, including perch *Perca fluviatilis*, stickleback *Gasterosteus aculeatus* and eel *Anguilla anguilla*, birds and dragonfly nymphs are known to feed on *G. tigrinus*. Fish can cause heavy mortality on *G. tigrinus* populations but, in turbid environments, *G. tigrinus* may escape heavy predation because benthivorous fish locate their prey visually (Kotta et al., 2010). *Dikerogammarus villosus*, another invasive gammarid, predates all size classes of *G. tigrinus*, with the smallest size classes suffering the highest predation (MacNeil et al., 2008).

In invaded waters, prevalence of parasites and epibionts (such as the rotifer *Embata parasitica* and the protozoan *Epistylis*) seems to be higher in the native *G. duebeni celticus* than in invading amphipods, including *G. tigrinus*, which may increase the species invasive success (Dunn and Dick, 1998). Fungus *Saprolegnia* spp. affects *G. tigrinus* (Dieleman and Pinkster, 1977, Van der Velde et al., 2000) and the parasite *Maritrema subdolum* (Trematoda) has been reported to occur in *G. tigrinus* in Poland (Rolbiecki and Normant, 2005)." (https://www.cabi.org/isc/datasheet/82074).

From Kotta et al. (2013): "*Gammarus tigrinus* appears to be competitively superior to the native gammarids, possibly leading to further decline of the native gammarid populations in the Gulf of Finland. "

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Gammarus tigrinus* has the capacity for a moderate range of natural dispersal in the RA area. It is a more sedentary and direct life-cycle species (i.e. no planktonic stage), hence it would not be expected to disperse widely. There is an adequate amount of supporting information suggesting that it may spread at a moderate rate in the RA area.

Background Information: Unlikely to disperse by algal rafting (Myers, 1993), *G. tigrinus* has been found to disperse by active migration in invaded ranges in Germany, Ireland and the Netherlands through canals and rivers (Pinkster et al., 1980).

From CABI: “In Europe, it spread quickly along the Baltic Sea south coast (at rates of around 40 km per year) and covering a total of about 1000 km from 1975 to 1998 (Grigorovich et al., 2005) (https://www.cabi.org/isc/datasheet/82074)”. It is unclear the degree to which this spread was aided by humans as it has also been introduced throughout its exotic range a number of times.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Gammarus tigrinus* has the capacity for a wide range of anthropogenic dispersal in the RA area. There seems to be solid evidence for wide-range dispersal through uptake in ballast and biofouling on vessels moving within the RA area.

Background Information: In Europe the spread is related to ballast water dispersal from the North American Coasts (Chambers, 1977, Ba et al., 2010). In Germany, *G. tigrinus* has been introduced as fish food because of its tolerance to saline and ion-rich polluted water (Van der Velde et al., 2000). It can be transported by ballast water (Chan et al., 2015, 2016). Ballast water transported by domestic coastal resupply and other boat traffic, could spread *G. tigrinus* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2015) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread. Ships visiting Churchill can carry live *Gammarus* spp. on their hulls (Chan et al., 2015) so biofouling of domestic supply vessels within the Hudson Bay region is also a vector for dispersal within the assessment region.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Gammarus tigrinus* would have a high impact in many areas on population growth in the RA area. Abundance of published information on impacts to populations, communities and coastal ecosystems from which impacts in the RA area can be extrapolated.

Background Information: Concurrent with the invasion of *G. tigrinus*, the diversity of native gammarids has declined (Herkül and Kotta, 2007).

From Kotta et al. (2013): “The species is competitively superior over native amphipods (MacNeil et al., 2003a) and is behaviourally aggressive towards native gammarids (Orav-Kotta et al., 2009). Co-occurrence of *G. tigrinus* with native gammarids increases exposure of native species to fish predation (Kotta et al., 2010). It is known to have replaced native communities of other amphipods such as *Gammarus zaddachi*, *Gammarus pulex* and *Gammarus duebeni* in Europe due to salinity tolerance and preference, life cycle, egg incubation period and time to reach sexual maturity (Pinkster et al., 1977).”

From Reisalu et al. (2016): “Irrespective of the competitive superiority of *G. tigrinus* over native gammarids, the invasive *G. tigrinus* does not monopolize the entire coastal area of the northern Baltic Sea but mostly outcompetes native species in its favoured habitats.”

From NOBANIS: “*Gammarus tigrinus* is able to outcompete many native gammarids in oligohaline waters. In Vistula Lagoon in the southern Baltic Sea, the native gammarids *G. duebeni* and *G. zaddachi* have been replaced by non-native species, predominantly *G. tigrinus* (Grabowski et al., 2006, Surowiec and Dobrzycka-Krahel, 2008). In any case the original amphipod fauna has been drastically changed in several places in northern Europe (Pinkster et al., 1992, Jazdzewski et al., 2004, Zettler, 2008). Life-history traits, such as early maturation, large brood size, and short generation time have been identified as possible reasons for the competitive superiority of *G. tigrinus* (Pinkster et al., 1977, Costello, 1993), though multivariate analyses showed that tolerance to changing salinity and pollution were better indicators for separating invasive and native species (Devin and Beisel, 2007, Grabowski et al., 2007a). Parasitism, both in native and invasive species, may also be involved in determining competitive success (MacNeil et al., 2003a), whereas differences in microhabitat preference and diel activity patterns have been demonstrated in cases of coexistence (MacNeil and Prenter, 2000, van Riel et al., 2007).” (https://www.nobanis.org/globalassets/speciesinfo/g/gammarus-tigrinus/gammarus-tigrinus.pdf)

From GLANSIS: "*Gammarus tigrinus* has colonized European waters extensively, spreading along the Baltic Sea coast at rates of around 40 km/year and covering a total of approximately 1000 km from 1975 to 1998 (Grigorovich et al., 2005). As a facultative carnivore of other macroinvertebrates, *G. tigrinus* is thought to influence community structure (e.g., trophic relationships) by monopolizing resources that would normally be consumed by its prey (Savage, 2000). The central European invasion of *G. tigrinus* has been accompanied by elimination of some native amphipod species from parts of the Rhine River, the Baltic Sea, and several waterbodies in the Netherlands. It is frequently a superior predator compared to native amphipods and could possibly have a reproductive advantage over such indigenous species as *G. duebeni*, *G. zaddachi*, and *G. pulex* (Pinkster et al., 1977, Grigorovich et al., 2005). Increased mortality in the Baltic Sea native amphipod, *Gammarus salinus*, has been attributed to increased competition with *G. tigrinus* over *Pilayella littoralis*, a mutually-grazed macrophyte species (Orav-Kotta et al., 2009). Furthermore, although *G. tigrinus* appeared to be a favored prey item of three-spined stickleback (*Gasterosteus aculeatus*) in the Baltic Sea, the presence of *G. tigrinus* also facilitated fish predation on *G. salinus* in certain habitat types (Kotta et al., 2010).

Gammarus tigrinus coexists in Ireland with the native opossum shrimp *Mysis relicta* and there is mutual predation (Bailey et al., 2006). However, the mysid has been forced to change its use of microhabitat, exposing itself to increased fish predation due to the presence of *G. tigrinus* (Bailey et al., 2006). *Gammarus tigrinus* also preys on a relatively small North American amphipod, *Crangonyx pseudogracilis*, in Ireland and could similarly prey on it in the Great Lakes (Dick, 1996, Grigorovich et al., 2005). However, while *G. tigrinus* can exclude *C. pseudogracilis* on habitats with good water quality, in poor water quality habitats, this may not be the case (MacNeil et al., 2001)."

(https://nas.er.usgs.gov/queries/greatLakes/FactSheet.aspx?SpeciesID=2650&Potential=N&Type=0&HUCNumber=DGreatLakes)

From CABI: “In areas where *G. tigrinus* and other non-native gammarids dominate, the original amphipod fauna has often been drastically changed (e.g. Pinkster et al., 1992, Jazdzewski et al., 2004, Zettler, 2008). In the Baltic Sea, for example, the native gammarid species of the Vistula Lagoon (*G. zaddachi* and *G. duebeni)*, were common in 1952 to 1956 and 1970. They have been gradually replaced by *G. tigrinus* and other non-indigenous gammarids, such as *Pontogammarus robustoides*, *Dikerogammarus haemobaphes* and *Obesogammarus crassus* (Żmudziński, 1957, Jażdżewski, 1975, Jazdzewski et al., 2004, Grabowski et al., 2006). In the Vistula Delta, the native species *G. duebeni*, *G. zaddachi*, *G. salinus* and *G. oceanicus*, common in 1967 to 1971, had mostly been replaced in 1998 to 2000 by the non-indigenous species *D. haemobaphes*, *P. robustoides* and *G. tigrinus* (Jażdżewski, 1975, Jazdzewski et al., 2004)). In 2008 to 2010, *G. tigrinus* was the most dominant species in nearshore zones of both the Vistula Lagoon and Vistula Delta (Dobrzycka-Krahel et al., 2013), while native gammarid species went totally extinct. Similarly, following the invasion of *G. tigrinus*, *G. salinus* almost disappeared from Koiguste Bay, Estonia (Herkül et al., 2009) and *G. duebeni* from Watch Lane Flash, UK (Savage, 1982). In Northern Ireland, *G. tigrinus* may be disadvantaged from intraguild predation by the native *G. duebeni celticus* than vice versa, but the native species appears more disadvantaged with respect to drift, parasitism and the interaction of the two (MacNeil et al., 2003b, MacNeil et al., 2003c).

One of the reasons native gammarids retreat is the fact that *G. tigrinus* proved to be competitively superior to the majority of native gammarids in Europe (Pinkster et al., 1992, Grabowski et al., 2006, Kotta et al., 2006b, Orav-Kotta et al., 2009, Kotta et al., 2010, Sareyka et al., 2011). Due to its ecological plasticity, aggressiveness and fast reproduction, *G. tigrinus* competes successfully for food and space with native gammarid species, forcing them to leave their preferred habitats and increasing exposure to fish predation (Orav-Kotta et al., 2009, Kotta et al., 2010, Kotta et al., 2011). For example, in the Gulf of Finland, the omnivorous *G. tigrinus* is able to outcompete and replace native herbivorous *Gammarus species* (Packalén et al., 2008). In the Rhine River, interspecific competitive stress caused by *G. tigrinus*, combined with physiological stress caused by the ion-enriched Rhine water, may have restricted the distribution of *G. pulex* to ion-poor ditches and brooks discharging into this river (van Riel et al., 2009). It is important to note that for some regions, drastic changes in salinity and not the introduction of *G. tigrinus*, are the most important factor causing native gammarid decline (Piscart et al., 2005, Boets et al., 2011, Kotta et al., 2014).

Being an active carnivore, *G. tigrinus* shows significant predatory impact on other macroinvertebrates. In Germany, the impact was noted by Fries and Tesch (1965). In the Netherlands Roos (1979) reported the species feeding on polyps of *Cordylophora caspia* and in Ireland *G. tigrinus* coexists with the native opossum shrimp *Mysis relicta* through mutual predation (Bailey et al., 2006). Females and recently moulted *M. relicta* are particularly vulnerable to predation by *G. tigrinus*, which has forced the species to change its microhabitat use, exposing itself more to fish predation.” (https://www.cabi.org/isc/datasheet/82074)

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Gammarus tigrinus* would have a high impact in many areas on communities in the RA area. An abundance of published information on impacts to populations, communities and coastal ecosystems from which impacts in the RA area can be extrapolated.

Background Information: From Kotta et al. (2013): “Experimental evidence indicates the species has a strong potential to modify benthic community structure and functioning in the whole coastal zone of the northern Baltic Sea (Orav-Kotta et al., 2009, Kotta et al., 2010, Kotta et al., 2011, Sareyka et al., 2011, Kotta et al., 2014). It has also been shown to alter the species composition of the local *Gammarus* assemblages in the Barents Sea (Packalén et al., 2008)."

From Reisalu et al. (2016): "Irrespective to the competitive superiority of *G. tigrinus* over the native gammarids, the invasive *G. tigrinus* does not monopolize the entire coastal area of the northern Baltic Sea but mostly outcompetes native species in its favoured habitats."

From NOBANIS: "*Gammarus tigrinus* is able to outcompete many native gammarids in oligohaline waters. In Vistula Lagoon in the southern Baltic Sea, the native gammarids *G. duebeni* and *G. zaddachi* have been replaced by non-native species, predominantly *G. tigrinus* (Grabowski et al., 2006, Surowiec and Dobrzycka-Krahel, 2008). In any case the original amphipod fauna has been drastically changed in several places in northern Europe (Pinkster et al., 1992, Jazdzewski et al., 2004, Zettler, 2008). Life-history traits, such as early maturation, large brood size, short generation time have been identified as possible reasons for the competitive superiority of *G. tigrinus* (Pinkster et al., 1977, Costello, 1993), though multivariate analyses showed that tolerance to changing salinity and pollution were better indicators for separating invasive and native species (Devin and Beisel, 2007, Grabowski et al., 2007a). Parasitism, both in native and invasive species, may also be involved in determining competitive success (MacNeil et al., 2003a), whereas differences in microhabitat preference and diel activity patterns have been demonstrated in cases of coexistence (MacNeil and Prenter, 2000, van Riel et al., 2007)." (https://www.nobanis.org/globalassets/speciesinfo/g/gammarus-tigrinus/gammarus-tigrinus.pdf).

From GLANSIS: "*Gammarus tigrinus* has colonized European waters extensively, spreading along the Baltic Sea coast at rates of around 40 km/year and covering a total of approximately 1000 km from 1975 to 1998 (Grigorovich et al., 2005). As a facultative carnivore of other macroinvertebrates, *G. tigrinus* is thought to influence community structure (e.g., trophic relationships) by monopolizing resources that would normally be consumed by its prey (Savage, 2000). The central European invasion of *G. tigrinus* has been accompanied by elimination of some native amphipod species from parts of the Rhine River, the Baltic Sea, and several waterbodies in the Netherlands. It is frequently a superior predator compared to native amphipods and could possibly have a reproductive advantage over such indigenous species as *G. duebeni*, *G. zaddachi*, and *G. pulex* (Pinkster et al., 1977, Grigorovich et al., 2005). Increased mortality in the Baltic Sea native amphipod, *G. salinus*, has been attributed to increased competition with *G. tigrinus* over *Pilayella littoralis*, a mutually-grazed macrophyte species (Orav-Kotta et al., 2009). Furthermore, although *G. tigrinus* appeared to be a favored prey item of three-spined stickleback (*Gasterosteus aculeatus*) in the Baltic Sea, the presence of *G. tigrinus* also facilitated fish predation on *G. salinus* in certain habitat types (Kotta et al., 2010).

Gammarus tigrinus coexists in Ireland with the native opossum shrimp *Mysis relicta* and there is mutual predation (Bailey et al., 2006). However, the mysid has been forced to change its use of microhabitat, exposing itself to increased fish predation due to the presence of *G. tigrinus* (Bailey et al., 2006). *Gammarus tigrinus* also preys on a relatively small North American amphipod, *Crangonyx pseudogracilis*, in Ireland and could similarly prey on it in the Great Lakes (Dick, 1996, Grigorovich et al., 2005). However, while *G. tigrinus* can exclude *C. pseudogracilis* from habitats with good water quality, in poor water quality habitats, this may not be the case (MacNeil et al., 2001)."
(https://nas.er.usgs.gov/queries/greatLakes/FactSheet.aspx?SpeciesID=2650&Potential=N&Type=0&HUCNumber=DGreatLakes)

From CABI: "In areas where *G. tigrinus* and other non-native gammarids dominate, the original amphipod fauna has often been drastically changed (e.g. Pinkster et al., 1992, Jazdzewski et al., 2004, Zettler, 2008). In the Baltic Sea, for example, the native gammarid species of the

Vistula Lagoon (*G. zaddachi* and *G. duebeni)*, were common in 1952 to 1956 and 1970. They have been gradually replaced by *G. tigrinus* and other non-indigenous gammarids, such as *Pontogammarus robustoides*, *Dikerogammarus haemobaphes* and *Obesogammarus crassus* (Żmudziński, 1957, Jażdżewski, 1975, Jazdzewski et al., 2004, Grabowski et al., 2006). In the Vistula Delta, the native species *G. duebeni*, *G. zaddachi*, *G. salinus* and *G. oceanicus*, common in 1967 to 1971, had mostly been replaced in 1998 to 2000 by the non-indigenous species *D. haemobaphes*, *P. robustoides* and *G. tigrinus* (Jażdżewski, 1975, Jazdzewski et al., 2004)). In 2008 to 2010, *G. tigrinus* was the most dominant species in nearshore zones of both the Vistula Lagoon and Vistula Delta (Dobrzycka-Krahel et al., 2013), while native gammarid species went totally extinct. Similarly, following the invasion of *G. tigrinus*, *G. salinus* almost disappeared from Koiguste Bay, Estonia (Herkül et al., 2009) and *G. duebeni* from Watch Lane Flash, UK (Savage, 1982). In Northern Ireland, *G. tigrinus* may be more disadvantaged from intraguild predation by the native *G. duebeni celticus* than vice versa, but the native species appears more disadvantaged with respect to drift, parasitism and the interaction of the two (MacNeil et al., 2003b, MacNeil et al., 2003c).

One of the reasons native gammarids retreat is the fact that *G. tigrinus* proved to be competitively superior to the majority of native gammarids in Europe (Pinkster et al., 1992, Grabowski et al., 2006, Kotta et al., 2006b, Orav-Kotta et al., 2009, Kotta et al., 2010, Sareyka et al., 2011). Due to its ecological plasticity, aggressiveness and fast reproduction, *G. tigrinus* competes successfully for food and space with native gammarid species, forcing them to leave their preferred habitats and increasing exposure to fish predation (Orav-Kotta et al., 2009, Kotta et al., 2010, Kotta et al., 2011). For example, in the Gulf of Finland, the omnivorous *G. tigrinus* is able to outcompete and replace native herbivorous *Gammarus species* (Packalén et al., 2008). In the Rhine River, interspecific competitive stress caused by *G. tigrinus*, combined with physiological stress caused by the ion-enriched Rhine water, may have restricted the distribution of *G. pulex* to ion-poor ditches and brooks discharging into this river (van Riel et al., 2009). It is important to note that for some regions, drastic changes in salinity and not the introduction of *G. tigrinus*, are the most important factor causing native gammarid decline (Piscart et al., 2005, Boets et al., 2011, Kotta et al., 2014).

Being an active carnivore, *G. tigrinus* shows significant predatory impact on other macroinvertebrates. In Germany, the impact was noted by Fries and Tesch (1965). In the Netherlands Roos (1979) reported the species feeding on polyps of *Cordylophora caspia* and in Ireland *G. tigrinus* coexists with the native opossum shrimp *Mysis relicta* through mutual predation (Bailey et al., 2006). Females and recently moulted *M. relicta* are particularly vulnerable to predation by *G. tigrinus*, which has forced the species to change its microhabitat use, exposing itself more to fish predation." (https://www.cabi.org/isc/datasheet/82074)

Q11- Impact on habitats **(Score = 2, Certainty = 2)**

*Score Rationale: S*ome reliable information suggests *Gammarus tigrinus* would have high impact in few areas or moderate impact in many areas on habitat in the RA area.

Background Information: G. tigrinus is known to clean the water of detritus and dead animals (Schmitz, 1960).

From CABI: "Although *G. tigrinus* can function as a detritivore and shred decaying leaf material, it is not able to compensate for the activity of native species, which replacement has led to a decrease of at least 66% in the rate of leaf litter recycling (Van der Velde et al., 2000, Piscart et al., 2011). Furthermore, despite being a food source for commercial fish when abundant, the species becomes included in the food web, causing changes and shifts in fish diet, which results in overall changes of the littoral food web (Kelleher et al., 1998, Kelleher et al., 2000, Grabowski et al., 2006). Recent experimental evidence indicates that *G. tigrinus* has a strong potential to modify benthic community structure and functioning in invaded coastal zones (Orav-Kotta et al., 2009, Kotta et al., 2010, Kotta et al., 2011, Sareyka et al., 2011, Kotta et al., 2013)" (https://www.cabi.org/isc/datasheet/82074)

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Gammarus tigrinus* would have a high impact in many areas on ecosystem function in the RA area. Reliable information shows this species impacts food webs and energy transfer within the ecosystem.

Background Information: From Kotta et al. (2013): "Experimental evidence indicates the species has a strong potential to modify benthic community structure and functioning in the whole coastal zone of the northern Baltic Sea (Orav-Kotta et al., 2009, Kotta et al., 2010, Kotta et al., 2011, Sareyka et al., 2011, Kotta et al., 2014).

From CABI: "Although *G. tigrinus* can function as a detritivore and shred decaying leaf material, it is not able to compensate for the activity of native species, which replacement has led to a decrease of at least 66% in the rate of leaf litter recycling (Van der Velde et al., 2000, Piscart et al., 2011). Furthermore, despite being a food source for commercial fish, when abundant, the species becomes included in the food web, causing changes and shifts in fish diet, which results in overall changes of the littoral food web (Kelleher et al., 1998, Kelleher et al., 2000, Grabowski et al., 2006). Recent experimental evidence indicates that *G. tigrinus* has a strong potential to modify benthic community structure and functioning in invaded coastal zones (Orav-Kotta et al., 2009, Kotta et al., 2010, Kotta et al., 2011, Sareyka et al., 2011, Kotta et al., 2013)." (https://www.cabi.org/isc/datasheet/82074)

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Gammarus tigrinus* would have low or no impact associated with diseases, parasites or travelers in the RA area. There is little evidence of risk in the RA area as hosts are not present there. Trematode (digenean) lifecycles require multiple correct hosts and some life stages are very host-specific (particularly snail stage). Eel parasites are also host-specific for eels.

Background Information: From CABI: "*G. tigrinus* invasions have also been accompanied by the simultaneous introduction of alien parasites (Leppäkoski et al., 2002). *G. tigrinus* has been shown to be an intermediate host to the acanthocephalan *Paratenuisentis ambiguus*, which definitive host is the American eel *Anguilla rostrata* (Samuel and Bullock, 1981, Taraschewski et al., 1987, Morozinska-Gogol, 2008). This acanthocephalan seems to have been introduced with *G. tigrinus* into Europe, where its final European host is *Anguilla anguilla* (Thielen, 2005, Taraschewski, 2006). During the 1990s, both exotics had spread into the Rhine River, with *G. tigrinus* dominating the amphipod fauna and *P. ambiguus* dominating the intestinal helminth community of local eels (Sures and Streit, 2001). However, after the inauguration of the Rhine-Main-Danube Canal in 1992 and subsequent invasion by the competitively superior *Dikerogammarus villosus*, *G. tigrinus* was eliminated and *P. ambiguus* stopped being the dominant intestinal helminth of the European eel, not being recorded since 2001 (Dick and Platvoet, 2000, Thielen, 2005). The parasite has recently been found in eels in Polish coastal waters (Morozinska-Gogol, 2008) and may also occur in other localities within *G. tigrinus'* introduced range." (https://www.cabi.org/isc/datasheet/82074)

There are no records of *Anguilla* species in the assessment area.

From NOBANIS: "*G. tigrinus* has been shown to be second intermediate host for a native digenean, *Maritrema subdolum* Jägerskiöld, 1909 (Rolbiecki and Normant, 2005). " (https://www.nobanis.org/globalassets/speciesinfo/g/gammarus-tigrinus/gammarus-tigrinus.pdf) The final host is a shorebird. Specificity of the first host (*Hydroibia* spp) is unclear (but trematodes are usually quite host specific for the first snail host). One related species (*Hydrobia* *minuta* - now known as *Ecrobia truncata*) has a distribution throughout the Arctic in Canada. Completion of such a complex lifecycle is unlikely, given the uncertainties of species matching.

From NOBANIS: "*G. tigrinus* is an intermediate host for the eel parasite *Paratenuisentis ambiguus* (Van Cleave, 1921) in Germany, but because only laboratory reared amphipods were introduced from England to Germany, it is unlikely that the acanthocephalan parasite was brought to Germany with the amphipod (Taraschewski et al., 1987). The parasite has recently been found in eels in Polish coastal waters (Morozinska-Gogol, 2008), and may also occur in other localities within the introduced range of *G. tigrinus*." (https://www.nobanis.org/globalassets/speciesinfo/g/gammarus-tigrinus/gammarus-tigrinus.pdf)

From CABI: "In invaded waters, prevalence of parasites and epibionts (such as the rotifer *Embata parasitica* and the protozoan *Epistylis*) seems to be higher in the native *G. duebeni celticus* than in invading amphipods, including *G. tigrinus*, which may increase the species invasive success (Dunn and Dick, 1998). Fungus (*Saprolegnia* spp.) affect *G. tigrinus* (Dieleman and Pinkster, 1977, Van der Velde et al., 2000) and the parasite *Maritrema subdolum* (Trematoda) has been reported to occur in *G. tigrinus* in Poland (Rolbiecki and Normant, 2005)." (https://www.cabi.org/isc/datasheet/82074)

Q14- Genetic impact (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Gammarus tigrinus* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native gammarids of the same genus in the RA area, but no evidence of hybridization found in the literature.

Background Information: Other *Gammarus* species exist in the region (*G. oceanicus*, *G. setosus*, and *G. wilkitzkii* and are widely distributed in the Hudson Bay LME (see also Atkinson and Wacasey, 1989a; North/South Consultants Inc. 2006). No evidence of hybridization between this and other species was found in the literature.

Q15- Impact on at-risk species (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Gammarus tigrinus* would have low or no impact on at-risk or depleted species in the RA area. There is good data to suggest that there should be little impact on deep-water and pelagic species.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. It is not likely to affect fish or mammals at risk in the assessment area as *Gammarus tigrinus* is quite coastal in its distribution and effects.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Gammarus tigrinus* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. Little impact on deeper-water and more pelagic species which are typically harvested in the RA area.

Background Information: Invertebrate invasion can impact fish population dynamics substantially if the fish cannot adapt their feeding behavior to accompanying changes in community structure (Kelly and Dick, 2005, and references therein). *Gammarus pulex* impacts brown trout (*Salmo trutta*) in European waters, altering population dynamics and the diet of juveniles (Kelly and Dick, 2005). Damaging effects on fishing gear and trapped fish have also been shown when *G. tigrinus* in high densities (Pinkster et al., 1977). Other salmonid fishes are harvested for subsistence in the assessment area (Stewart and Lockhart, 2005).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Gammarus tigrinus* is invasive somewhere else in the world. Solid evidence of establishment and impacts in invaded areas.

Background Information: From Ba et al. (2010): “Widespread species that occurs principally in estuaries of the northwestern Atlantic and is distributed from the St. Lawrence River in Quebec to Florida (Bousfield, 1958, Bousfield, 1973). It was introduced into British waters by ballast water in 1931 (Sexton, 1939), then spread to similar habitats in Western Europe and, most recently, to Eastern Europe, the Baltic Sea, and the Laurentian Great Lakes (Kelly et al., 2006b). Dispersion in the last decade was extremely rapid in Eastern Europe. *Gammarus tigrinus* is not only restricted to near port water bodies, but is also widespread in inland water systems. In Europe, *G. tigrinus* might have invaded regions of Baltic Sea rapidly because of lower seawater salinity (10 PSU) and it was deliberately introduced in Germany as fish food (Van der Velde et al., 2000). It has also been found in the Gulf of Paria and Orinoco Delta in Venezuela (latitude 10°N) (Capelo et al., 2004).”

Pontogammarus robustoides **G.O. Sars, 1894**

Phylum: Arthropoda

Class: Malacostraca

Order: Amphipoda

Family: Gammaridae

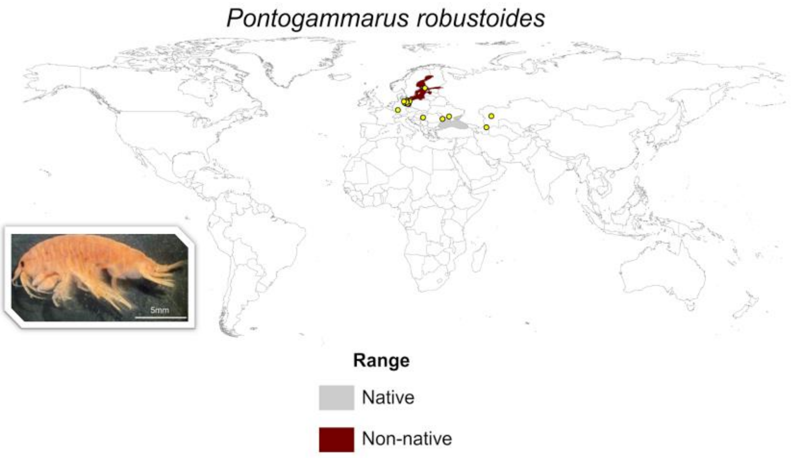

Figure 4: Ecoregions where Pontogammarus robustoides is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Pontogammarus robustoides occurrence points were harvested from GBIF.org (https://doi.org/10.15468/dl.44sc8y, 14 December 2020). Note that this species can also be present in freshwater ecosystems. Hence inland points were left on the map. Picture of P. robustoides modified from https://www.cabi.org/isc/datasheet/119602

CMIST scores for *P. robustoides:*

Mean adjusted Likelihood of Invasion: 1.80

Mean adjusted Impact of Invasion: 2.06

Mean adjusted Overall CMIST score: 3.70

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There are no reports of *P. robustoides* in Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016, Dispas, 2019).

Q2- Rate of introduction (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* could arrive infrequently in low numbers of individuals in the RA area. There are limited vectors for transport and it is not found in adjacent regions.

Background Information: Pontogammarus robustoides is present in ports that are directly connected by shipping to Churchill and could potentially be entrained and transported by these vessels in ballast water, or on fouled hulls (Chan et al., 2012).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Pontogammarus robustoides.* Only a moderate proportion of coastal areas have mud/sand substrate and shallow embayments; there is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: From CABI: “In its native range, the species has been found in brackish and freshwater bays, coastal lakes and lagoons, lower courses and estuaries. It is usually found in the open shallow waters among macrophytes growing on a sand or silted sand bottom (Yemelianova, 1994). Juvenile and adult gammarids exhibit different habitat preferences. Adults do not discriminate between artificial and natural substrata, or among most of the tested species of plants (*Myriophyllum spicatum*, *Ceratophyllum demersum*, *Potamogeton perfoliatus*, *Elodea canadensis*), whereas juveniles prefer all tested macrophytes over artificial substrata and prefer plants with finer leaf elements, i.e. *M. spicatum* and *C. demersum* over the other plants and *E. canadensis* over *P. perfoliatus* (Czarnecka et al., 2010). The indiscrimination of adult stages regarding substrate may also explain why the species, considered to be strictly phytophilous, is often found on stony or sand-muddy bottom. It is resistant to substratum drying and can bury into the substratum. This ability allows the species to invade and persist in habitats which experience water level fluctuations.”(https://www.cabi.org/isc/datasheet/119602)

Q4- Survival (suitable environment) (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest the RA area offers a negligible proportion of suitable environmental conditions for *Pontogammarus robustoides*. Most of the RA area has higher salinity values. Temperatures in much of the RA area are below 0° C, the minimum known temperature within its native range. There is limited information on the temperature tolerance of the species and minimum temperature thresholds have been estimated based on known distribution only.

Background Information: From CABI: “*Pontogammarus robustoides* can tolerate a wide range of salinities and displays flexible osmotic capacity: from 0 to 23 PSU (Dobrzycka-Krahel and Surowiec, 2011). Laboratory investigations have shown that this species can survive in fully saline sea water (up to 34 PSU) during 48hs in a flow-through experiment of ballast water

(Santagata et al., 2008). Chekunova (1960) revealed the importance of K and Ca metabolism on the survival of the species. Berezina and Panov (2003) found that the concentration Na^+ should be at least 17 mg/L for successful reproduction of *P. robustoides*; however, the species has been found in Lithuanian lakes with Na+ as low as 3.4 mg/L (Arbačiauskas and Gumuliauskaitė, 2007). Effect of temperature on *P. robustoides* has not yet been studied in detail. In its native habitat water temperature may vary from 0 to 30 ºC (Mordukhai-Boltovskoi, 1960). Kurashov et al. (2012) noted that increasing average annual temperatures of upper water levels in European lakes (such as Lake Lagoda) in the 20th century favours their invasion by Ponto-Caspian species." (https://www.cabi.org/isc/datasheet/119602)

From Arbačiauskas and Gumuliauskaitė (2007) and references therein: "Although described as a euryhaline freshwater species or as a brackish water species it establishes sustainable populations only in salinities not exceeding 3–4 PSU (e.g., characteristic of Baltic lagoons). There is a hypothesis that in higher latitudes where stagnant waters are ice-covered for a substantial portion of the year, oxygen content in the water during the winter may be a decisive factor for the long-term survival of *P. robustoides* (Arbačiauskas, 2002, Arbačiauskas, 2005)."

Q5- Establishment (reproductive requirements) (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available which suggests reproductive requirements are almost never available for *Pontogammarus robustoides* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Most of the RA area has higher salinity values. Low salinity regions likely represent a very small proportion of the RA area. There is no specific data about temperature, and K or Ca layers are not available. Some temperature information was found regarding temperatures required for spawning, however given lack of information on the impact of salinity on reproduction success and the fact that populations do not seem to be established at PSU over 3 or 4, this suggests that the species is not likely to establish populations in a wide part of the Hudson Bay Complex.

Background Information: From CABI: "The life cycle of *P. robustoides* consists of direct development with no independent larval stage. Females carry their embryos in a brood chamber between the pereopods. When released, the juveniles reach maturity after several molts, without any metamorphosis. Chekunova (1960) revealed the importance of K and Ca metabolism on the survival of the species. Berezina and Panov (2003) found that the concentration Na^+ should be at least 17 mg/L for successful reproduction of *P. robustoides* (Berezina and Panov, 2003)". (https://www.cabi.org/isc/datasheet/119602)

From Berezina 2016: "Reproduction has been shown to be initiated at 8-10ºC in spring and cease at 5ºC in fall. Although, no specific information was found about temperature and salinity tolerance of embryos & juveniles, studies show that rate of development increases with temperature in the range tested (13-25 ºC) (Ioffe and Maximova, 1968); general data from survival (Question 4) is assumed to represent salinity/temperature limits for embryo and juvenile development since the species has direct development."

From Arbačiauskas and Gumuliauskaitė (2007) and references therein: "In its native habitat water temperature may vary from 0 to 30 ºC (Mordukhai-Boltovskoi, 1960) although described as a euryhaline freshwater species or as a brackish water species it only establishes sustainable populations in salinities not exceeding 3–4 PSU."

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Pontogammarus robustoides*. There may be closely related species in the RA that could act as predators, but no indication that these can severely limit population growth; the only species that may be able to limit growth is an invasive species that is not found in the RA area; likewise, parasites in the native habitat are found to have low infestation rates and unlikely to be present in the RA area. There is limited information on specific predators and parasites with respect to how they may regulate population size, particularly in the RA area.

Background Information: From CABI: "Several microparasites of invasive gammarids in Polish waters have been recognized (Ovcharenko et al., 2009, Ovcharenko and Yemeliyanova, 2009). Four species of gregarines (*Uradiophora ramosa*, *U. longissima*, *Cephaloidophora similis*, *C. mucronata*) and five microsporidians (*Nosema dikerogammari*, *N. pontogammari*, *Thelohania* sp. 2, *Thelohania* sp. 5; *Pleistophora muelleri*) were associated with hosts of Ponto-Caspian origins. Infestation rates did not exceed 3%. The authors did not register any transition of parasites of the Ponto-Caspian hosts to the hosts of native fauna." (https://www.cabi.org/isc/datasheet/119602)

The growth of *P. robustoides* was experimentally shown to be reduced in the presence of predators (Jermacz et al., 2017). The invasive *Dikerogammarus villosus* is recognized as being capable of eliminating other gammarids, both native and exotic, including *P. robustoides* (Dick and Platvoet, 2000{Dick, 2002 #1613)}.

From GLANSIS: "When abundant, *P. robustoides* significantly contributes to the diets of various fish species (Bubinas, 1979, Arbačiauskas and Gumuliauskaitė, 2007), but the extent to which this predation will have an effect on potential populations in the Great Lakes is unknown." (https://nas.er.usgs.gov/queries/greatlakes/FactSheet.aspx?SpeciesID=24&Potential=Y&Type=2&HUCNumber)

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* has the capacity for a moderate range of natural dispersal in the RA area. There is moderate evidence available to suggest a reasonable dispersal range via migration/crawling on bottom, however long range dispersal may be limited since the species does not have a pelagic life stage.

Background Information: From CABI: "The species is able to disperse naturally, and dispersal in rivers and lakes may occur both upstream and downstream. This ability to naturally disperse through freshwater streams might contribute to the distribution of this species through rivers and channel routes in Europe (Arbačiauskas et al., 2011a). *P. robustoides* exhibits high dispersal potential and is constantly expanding its range of distribution. *P. robustoides* disperses both naturally and via human activity. The species is able to disperse naturally, as was shown by Arbačiauskas et al., (2011a) in the Neman River basin." (https://www.cabi.org/isc/datasheet/119602)

From Arbačiauskas and Gumuliauskaitė (2007) and references therein: "A few possible methods of invasion of *P. robustoides* into the deltas of the Vistula and Oder rivers have been suggested (Gruszka, 1999, Jażdżewski and Konopacka, 2000, Bij de Vaate et al., 2002, Jazdzewski et al., 2004). Firstly, transmission of this pontogammarid from the Curonian Lagoon via ballast waters; secondly, dispersal through the coastal waters of the Baltic Sea; and thirdly, a freshwater route from the Nemunas River basin via the Pregel River system which provides a direct connection between the Curonian and Vistula lagoons. As *P. robustoides* also was detected in the lower Vistula reaches, the central invasion corridor, i.e. the route connecting the Dnieper and Vistula basins via Pripet-Bug canal, is also under consideration."

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Pontogammarus robustoides* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is strong evidence of dispersal due to transport in ballast and via hull fouling.

Background Information: From CABI: “Anthropogenic factors facilitate the spread of *P. robustoides* along large rivers (Vistula, Oder, Neman, Elbe) and navigable canals and into artificial reservoirs as well as lakes (Grabowski, 2011). It can penetrate inland waters through shipping (Reinhold and Tittizer, 1997, Reinhold and Tittizer, 1999, Jazdzewski et al., 2002, Grabowski and Bącela, 2005). *Pontogammarus robustoides* can be transferred not only in ballast waters but also in the hull fouling of ships, especially over short distances by slow-speed vessels, such a dispersal method being possibly the most important (Arbačiauskas and Gumuliauskaitė, 2007). There has also been intentional release in water reservoirs and lakes as fish-food (Arbačiauskas et al., 2010, Grabowski, 2011).

Pontogammarus robustoides exhibits high dispersal potential and is constantly expanding its range of distribution. *Pontogammarus robustoides* disperses both naturally and by human activity. It seems to have expanded in the Baltic Sea via jump dispersal, clearly suggesting strong involvement of anthropogenic factors (Arbačiauskas and Gumuliauskaitė, 2007).” (https://www.cabi.org/isc/datasheet/119602)

Hull fouling and ballast transported by domestic coastal resupply and other boat traffic, could spread *P. robustoides* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

From Arbačiauskas and Gumuliauskaitė (2007) and references therein: “A few possible methods of invasion of *P. robustoides* into the deltas of the Vistula and Oder rivers have been suggested (Gruszka, 1999, Jażdżewski and Konopacka, 2000, Bij de Vaate et al., 2002, Jazdzewski et al., 2004). Firstly, transmission of this pontogammarid from the Curonian Lagoon via ballast waters; secondly, dispersal through the coastal waters of the Baltic Sea; and thirdly, a freshwater route from the Nemunas River basin via the Pregel River system which provides a direct connection between the Curonian and Vistula lagoons. As *P. robustoides* also was detected in the lower Vistula reaches, the central invasion corridor, i.e. the route connecting the Dnieper and Vistula basins via Pripet-Bug canal, is also under consideration.”

Q9- Impact on population (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* would have a high impact in many areas on population growth in the RA area. Many impacts have been shown but much evidence is from extrapolations from more freshwater environments.

Background Information: From CABI: “Reductions in species richness (1.5- to 1.6-fold), in community diversity (more than two-fold) and in the biomass of indigenous invertebrates (excluding chironomids, which exhibited high lake-specific biomass variation) have been found in the places with well-established *P. robustoides* populations (Gumuliauskaitė and Arbačiauskas, 2008). A decline of native *G. zaddachi* and *G. duebeni* in the Baltic Sea was reported parallel to the appearance of *P. robustoides* and other alien amphipods (Jazdzewski et al., 2004, Ezhova et al., 2005, Grabowski et al., 2006). Apart from its influence on local fauna, *P. robustoides* also affects algae biomass (e.g., decrease of in littoral zones of *Cladophora* lakes after the established population of *P. robustiodes* (Gasjunas, 1972).

In many locations the activity of *P. robustoides* has had a significant impact on biodiversity. Berezina and Panov (2003) described the favourable impact of *P. robustoides* on food webs. By intensively consuming plant food, *P. robustiodes* produces abundant faeces which increase organic matter availability for benthic detritivores. In the Gulf of Finland, for example, at gammarid densities of 500-3000 ind/m(-2), the densities of detritivores were 2-3 times as high.

However, such activity also causes severe biocontamination, which was demonstrated by Arbačiauskas et al., (2011b) for Lithuanian rivers.

With increasing gammarid densities, the replacement of previously co-existing species and shifts in the densities of native invertebrates occurs. Orlova et al. (2006) believed that *P. robustoides* has strongly affected the community structure in the eastern Gulf of Finland since 1998.

Gumuliauskaitė and Arbačiauskas (2008) observed a detrimental impact of the species upon the native isopod *Asellus aquaticus* and a negative correlation with most of the higher taxa of native invertebrates. Arbačiauskas (2005) and Arbačiauskas (2008) described a decline of *Gammarus lacustris* and other native gammarids in Polish freshwaters as a result of adventive gammarid species. According to Surowiec and Dobrzycka-Krahel (2008) long-term studies in the Vistula Delta and Lagoon have shown a dramatic decline in the native gammarid species *Gammarus duebeni*, *G. zaddachi*, *G. salinus*, *G. oceanicus* and *G. varsoviensis* and the complete replacement of *Chaetogammarus ischnus* by other non-indigenous gammarids.

Pontogammarus robustoides may also affect macroinvertebrates in a specific way. Kobak et al. (2012) discovered that Ponto Caspian gammarids within mussel colonies have the capacity to compromise the normal functioning of bivalves by inducing responses in them similar to their anti-predator defenses. The most likely factor causing these changes was mechanical irritation of their soft parts by amphipod appendages.

Apart from its influence on local fauna, *P. robustoides* also affects algae biomass. Gasiūnas (1972) showed that an established population of *P. robustoides* in littoral zones of Lithuanian lakes contributed to the disappearance of *Cladophora* spp. in five years. Berezina and Golubkov (2008) hypothesized that the alien amphipods *Gmelinoides fasciatus* and *P. robustoides* are able to control macroalgae biomass in the eastern Gulf of Finland. The data obtained clearly indicated that the grazing amphipods can have a dramatic impact on *Cladophera glomerata* in the littoral zone of the eastern Gulf of Finland, and perhaps influence the macroalgal biomass when their populations are dense.

Generally, there is increasing evidence to suggest that *P. robustoides* has an adverse impact on the richness, biodiversity and biomass of native littoral assemblages. Possible reasons for the observed effects are increasing pollution and eutrophication, accompanied by competition between native and the alien species. The selective predation of invasive amphipods on native invertebrates is also considered to be a main impact mechanism." (https://www.cabi.org/isc/datasheet/119602)

From NOBANIS: "It is an aggressive amphipod species, known to prey on other organisms (e.g., Oligochaeta, Chironomidae) posing a potential threat to local benthic fauna. This may pose a threat to local benthic fauna, including native amphipods (e.g. *Gammarus lacustris* in freshwaters). In the brackish Vistula Lagoon a decline of native *Gammarus zaddachi* and *Gammarus duebeni* was reported parallel to the appearance of *Pontogammarus robustoides* and other alien amphipods (Jazdzewski et al., 2004). Yet, the true nature of this phenomenon is unknown. In habitats of Lithuanian inland waters where *P. robustoides* is well established and numerous, it significantly reduces species richness and community diversity (Gumuliauskaitė and Arbačiauskas, 2008)." https://www.nobanis.org/globalassets/speciesinfo/p/pontogammarus-robustoides/pontogammarus_robustoides.pdf

Q10- Impact on communities (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* would have high impact in many areas on communities in the RA area. Many impacts have been shown, but much evidence is from extrapolations from more freshwater environments.

Background Information: From Gumuliauskaitė and Arbačiauskas (2008): “In lake habitats where *P. robustoides* is well established and numerous, the newcomer significantly reduces species richness and community diversity (Gumuliauskaitė and Arbačiauskas, 2008). A decline of more than two-fold in littoral community diversity may occur, like in Lithuanian Lakes (Gumuliauskaitė and Arbačiauskas, 2008). Negative correlations between *P. robustoides* and most of the higher taxa of native invertebrates suggest that the invader may, directly or indirectly, affect various benthic animals. Invading amphipods have been shown to be capable of impacting negatively on the abundance and biomass of different indigenous invertebrates (Kelly et al., 2003). Given that it is omnivorous, the alien *P. robustoides* strongly affects the community structure with its versatile diet (Orlova et al., 2006).”

From CABI: “In many locations the activity of *P. robustoides* has had a significant impact on biodiversity. Berezina and Panov (2003) (described the favourable impact of *P. robustoides* on food webs. By intensively consuming plant food, *P. robustiodes* produces abundant faeces which increase organic matter availability for benthic detritivores. In the Gulf of Finland, for example, at gammarid densities of 500-3000 ind/m(-2), the densities of detritivores were 2-3 times as high. However, such activity also causes severe biocontamination, which was demonstrated by (Arbačiauskas et al., 2011b) for Lithuanian rivers.

With increasing in gammarid densities, the replacement of previously co-existing species and shifts in the densities of native invertebrates occurs. (Orlova et al., 2006) believed that *P. robustoides* has strongly affected the community structure in the eastern Gulf of Finland since 1998.

Gumuliauskaitė and Arbačiauskas (2008) observed a detrimental impact of the species upon the native isopod *Asellus aquaticus* and a negative correlation with most of the higher taxa of native invertebrates. Arbačiauskas (2005), (Arbačiauskas, 2008) described a decline of *Gammarus lacustris* and other native gammarids in Polish freshwaters as a result of adventive gammarid species. According to Surowiec and Dobrzycka-Krahel (2008) long-term studies in the Vistula Delta and Lagoon have shown a dramatic decline in the native gammarid species *Gammarus duebeni*, *G. zaddachi*, *G. salinus*, *G. oceanicus* and *G. varsoviensis* and the complete replacement of *Chaetogammarus ischnus* by other non-indigenous gammarids.

P. robustoides may also affect macroinvertebrates in a specific way. Kobak et al. (2012) discovered that Ponto Caspian gammarids within mussel colonies have the capacity to compromise the normal functioning of bivalves by inducing responses in them similar to their anti-predator defenses. The most likely factor causing these changes was mechanical irritation of their soft parts by amphipod appendages.

Apart from its influence on local fauna, *P. robustoides* also affects algae biomass. Gasiūnas (1972) showed that an established population of *P. robustoides* in littoral zones of Lithuanian lakes contributed to disappearance of *Cladophora spp.* in five years. Berezina and Golubkov (2008) hypothesized that the alien amphipods *Gmelinoides fasciatus* and *P. robustoides* are able to control macroalgae biomass in the eastern Gulf of Finland. The data obtained clearly indicated that the grazing amphipods can have a dramatic impact on *Cladophera glomerata* in the littoral zone of the eastern Gulf of Finland, and perhaps influence the macroalgal biomass when their populations are dense.

Generally, there is increasing evidence to suggest that *P. robustoides* has an adverse impact on the richness, biodiversity and biomass of native littoral assemblages. Possible reasons for the observed effects are increasing pollution and eutrophication, accompanied by competition between native and the alien species. The selective predation of invasive amphipods on native invertebrates is also considered to be a main impact mechanism.” (https://www.cabi.org/isc/datasheet/119602)

From NOBANIS: “It is an aggressive amphipod species, known to prey on other organisms (e.g., Oligochaeta, Chironomidae) posing a potential threat to local benthic fauna. This may pose a threat to local benthic fauna, including native amphipods (e.g. *G. lacustris* in freshwaters). In the brackish Vistula Lagoon a decline of native *G. zaddachi* and *G. duebeni* was reported parallel to the appearance of *Pontogammarus robustoides* and other alien amphipods (Jazdzewski et al., 2004). Yet, the true nature of this phenomenon is unknown. In habitats of Lithuanian inland waters where *P. robustoides* is well established and numerous, it significantly reduces species richness and community diversity (Gumuliauskaitė and Arbačiauskas, 2008).” (https://www.nobanis.org/globalassets/speciesinfo/p/pontogammarus-robustoides/pontogammarus_robustoides.pdf)

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Pontogammarus robustoides* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. Some impacts may be possible and based mainly on documented information from brackish systems which would be found in the RA area.

Background Information: From CABI: “*Pontogammarus robustoides* is omnivorous, but it can have an impact through grazing. It can dramatically impact *Cladophora glomerata* in littoral zones (e.g. Gulf of Finland) and perhaps control macroalgal biomass, changing the environment (Berezina et al., 2005). Berezina and Golubkov (2008) demonstrated that the alien amphipods *Gmelinoides fasciatus* and *P. robustoides* can have a dramatic impact on *C. glomerata* in the littoral zone of the eastern Gulf of Finland, and perhaps influence the macroalgal biomass when their populations are dense.” (https://www.cabi.org/isc/datasheet/119602)

From GLANSIS: “*P. robustoides* may affect the composition and abundance of littoral macrophytes through heavy grazing pressure, as seen with the Great Lakes nuisance algae *Cladophora* in the eastern Gulf of Finland (Berezina et al., 2005). Populations in this study were able to consume 4-5 g (dry weight) *Cladophora*/day”. (https://nas.er.usgs.gov/queries/greatlakes/FactSheet.aspx?SpeciesID=24&Potential=Y&Type=2&HUCNumber)

Q12- Impact on ecosystem function (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Pontogammarus robustoides* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. Some impacts may be possible and based mainly on documented information from brackish systems which would be found in the RA area.

Background Information: From CABI: “In the Baltic Sea, it provided a food base for many local fish species (perch) and also for invasive fish species such as racer goby *Neogobius gymnotrachelus* and monkey goby *Neogobius fluviatilis*. It can have a favorable impact on food webs by intensively consuming plant food and producing abundant faeces which increase organic matter availability for benthic detritivores. However, such activity can also cause severe biocontamination (Arbačiauskas et al., 2011a) for Lithuanian rivers.

“*Pontogammarus robustoides* is omnivorous, but it can have an impact through grazing. It can dramatically impact on *Cladophora glomerata* in littoral zones (e.g. Gulf of Finland) and perhaps control macroalgal biomass, changing the environment (Berezina et al., 2005). Berezina and Golubkov (2008) demonstrated that the alien amphipods *Gmelinoides fasciatus* and P. robustoides are able to control macroalgae biomass in the eastern Gulf of Finland. The data obtained clearly indicated that the grazing amphipods can have a dramatic impact on C. glomerata in the littoral zone of the eastern Gulf of Finland, and perhaps influence the macroalgal biomass when their populations are dense.” (https://www.cabi.org/isc/datasheet/119602)

From NOBANIS: “Generally, there is increasing evidence to suggest that *P. robustoides* has an adverse impact on the richness, biodiversity and biomass of native littoral assemblages. Possible reasons for the observed effects are increasing pollution and eutrophication, accompanied by competition between native and the alien species. The selective predation of invasive amphipods on native invertebrates is also considered to be a main impact mechanism.” (https://www.nobanis.org/globalassets/speciesinfo/p/pontogammarus-robustoides/pontogammarus_robustoides.pdf)

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Pontogammarus robustoides* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. The species is known to carry several microparasites that can infect fish, with high possibility of being transferred with a host, hence, there is the possibility that some could infect native fish species in the RA area. However, the host specificity and impacts are highly uncertain.

Background Information: From CABI: “Several microparasites *Uradiophora ramosa, U. longissima, Cephaloidophora similis, C. mucronata*, *Nosema dikerogammari, N. pontogammari, Thelohania sp. 2, Thelohania sp. 5 and Pleistophora muelleri* were associated with hosts of Ponto-Caspian origins. There is no register of any transition to hosts of native fauna (Ovcharenko et al., 2009, Ovcharenko and Yemeliyanova, 2009).

P. robustoides may serve as a vector for microparasites; however, at present there are no reports on the possible transfer of pathogens to native species.” (https://www.cabi.org/isc/datasheet/119602)

From NOBANIS: “As other gammarids, the species may be a vector of alien parasites and (may) transfer them to local fish species. (e.g. Trematoda and Acanthocephala) (Sulgostowska and Vojtková, 1992).” (https://www.nobanis.org/globalassets/speciesinfo/p/pontogammarus-robustoides/pontogammarus_robustoides.pdf, https://nas.er.usgs.gov/queries/greatlakes/FactSheet.aspx?SpeciesID=24&Potential=Y&Type=2&HUCNumber)

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Pontogammarus robustoides* would have low or no genetic impact on other species in the RA area. No related species of the same genus are in the RA area.

Background Information: There are no related native species occurring in the Hudson Bay Complex region or the Canadian Arctic and hybridization is therefore not possible (https://www.gbif.org, https://obis.org/, (Stewart and Lockhart, 2005).

Q15- Impact on at-risk species (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* would have low or no impact on at-risk or depleted species in the RA area. There is scant habitat overlap with species at risk in the RA area and therefore likely little effect.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and at-risk fish and mammal species are unlikely to be affected due to low habitat overlap with this nearshore brackish/freshwater species.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Pontogammarus robustoides* would have high impact in few areas or moderate impact in many areas on fished species in the RA area.

Possible impacts to fisheries species however, are likely to be moderate given that the species is brackish/freshwater and there is not great certainty of this. Impacts to fisheries/aquaculture species are likely not great as aquaculture is not really practiced in the area.

Background Information: From CABI: "Kobak et al. (2012) discovered that Ponto Caspian gammarids within mussel colonies have the capacity to compromise the normal functioning of bivalves by inducing responses in them similar to their anti-predator defenses. The most likely factor causing these changes was mechanical irritation of their soft parts by amphipod appendages." (https://www.cabi.org/isc/datasheet/119602)

Shallow water benthic species could be impacted by competition/food web effects and/or habitat/ecosystem effects (see responses to Q.9-12).

Mya spp., as well as Mussels (*Mytilus* spp.), sea urchin (*Strongylocentrotus droebachiensis*), brown sea cucumber (*Cucumaria japonica*) Iceland scallops (*Chlamys islandica*) are harvested by Inuit in the RA and could be impacted (Stewart and Lockhart, 2005); (Igloolik Hunters and Trappers, personal communication) if they overlap in brackish habitats. This is only likely for *Mya* spp., as well as Mussels (*Mytilus* spp.).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Pontogammarus robustoides* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: The native range covers Russia, Turkey, the Caucasus, Romania, Bulgaria and Ukraine territories (https://www.cabi.org/isc/datasheet/119602). It invaded the southern Baltic Sea in 1960 when it was introduced from the Black Sea basin to the Kaunas Reservoir (Neman River) and subsequently further north in Lithuania, Latvia and Russia. By the end of 20th century *Pontogammarus robustoides* had spread in the estuaries and along the course of several large rivers (e.g., Vistula, Oder/Odra, Neman/Nemunas), and also penetrated several lakes and reservoirs within their basins. During the last few decades it was recorded in the eastern part of the Gulf of Finland and along the Estonian coast (Strode et al., 2013 and references therein).

ZOOBENTHOS – BARNACLES

Amphibalanus amphitrite (Darwin, 1854)

- Phylum: Arthropoda
  - Class: Hexanauplia
    - Order: Sessilia
      - Family: Balanidae

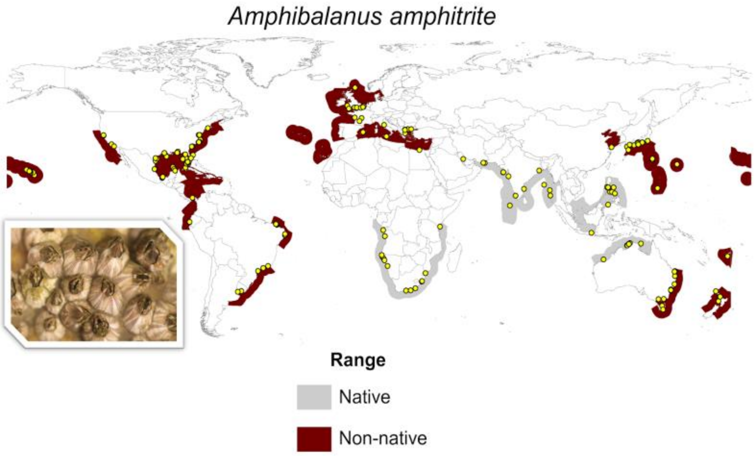

Figure 5: Ecoregions where Amphibalanus amphitrite is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) which have not been collected in an exhaustive manner. Amphibalanus amphitrite occurrence points were obtained from OBIS (https://obis.org/) and the African Register of Marine Species (AfResMaS) (http://www.marinespecies.org/afremas/aphia.php?p=taxdetails&id=421137). Picture of A. amphitrite modified from https://invasions.si.edu/nemesis/species_summary/89616

CMIST scores for *A. amphitrite:*

Mean adjusted Likelihood of Invasion: 2.15

Mean adjusted Impact of Invasion: 1.57

Mean adjusted Overall CMIST score: 3.38

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus amphitrite* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016) but larvae are being introduced in Churchill through ballast water, and visiting ships hulls are fouled with adults (Chan et al., 2015). No reports were found of *A. amphitrite* in Atlantic Canada. It was found for the first time in British Columbia via subtidal monitoring (establishment unknown) (Therriault et al., 2011).

Q2- Rate of introduction (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* could arrive frequently in high numbers in the RA area. The species is present in connected ports and there is clear evidence of arrival from multiple studies and vectors of transport.

Background Information: Ships visiting Churchill frequently carry live *A. amphitrite* on their hulls (5 of 13 ships tested; averaging 112,859 ± 69,678 individuals/ship) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ±300,133 ind/ship (Chan et al., 2015). Exposure occurs annually during the open water shipping season. *Amphibalanus amphitrite* is unlikely to be transferred via this route with live commercial shellfish.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Amphibalanus amphitrite.* There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem type data layers to support habitat classification. A moderate proportion of the RA area has rocky substrate.

Background Information: Amphibalanus amphitrite is characteristic of sheltered marine intertidal and shallow subtidal habitats where it attaches to a wide range of hard surfaces including docks, ship's hulls, logs, rocks, bivalve shells (Henry and McLaughlin, 1975, Grizel and Héral, 1991, Fofonoff et al., 2003) and crustacean carapaces (Farrapeira and Calado, 2010). It typically settles in the intertidal zone in tropical waters (e.g., Taiwan) and subtidally or on the bottoms of ships in temperate waters (Henry and McLaughlin, 1975). It survives on buoys in open coastal waters along the Belgian coast (Kerckhof and Cattrijsse, 2001). Lack of shelter from waves and the depth of ice scour are likely to limit the suitability of tidal and shallow subtidal habitats.

Q4- Survival (suitable environment) (Score = 1, Certainty = 3)

Score Rationale: Considerable amount of reliable information suggests the RA area offers a negligible proportion of suitable environmental conditions for *Amphibalanus amphitrite.* Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Amphibalanus amphitrite is a eurythermal species typically found in tropical to warm temperate waters. Low winter temperature may be a limiting factor responsible for cessation of recruitment (Qiu and Qian, 1999). Cirril activity was observed between 6°C and 37°C (Southward, 1957). Further work is needed using gradual acclimation to clarify the species' tolerance to low temperature at different salinities.

These barnacles are mesohaline to euryhaline. The lower salinity limit for *A. amphitrite* seems to be ~10 PSU, below which it is unable to osmoregulate (Qiu and Qian, 1999). In the laboratory, all adults died at ~5 PSU, and detrimental effects could still be found at 10 PSU, including lower survivorship in the 1st wk of the experiment, fewer molts, and lower percentage of individuals possessing ovaries and embryos. *Amphibalanus amphitrite* tolerates hypersaline conditions for short periods (100 g/L or ~100 PSU for 4 days) but showed marked declines in abundance at salinities as low as 50 g/L (~50 PSU) (Simpson and Hurlbert, 1998).

Adults and larvae can tolerate hypoxia and anoxia for a limited period (Desai and Prakash, 2009). Tolerance decreases if larvae are starved.

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Amphibalanus amphitrite* in the RA area. The areal extent of the maximum sea surface temperature >=10°C (the lower limit for reproduction/larval development), calculated using ArcGIS, found a moderate amount of suitable habitat in the RA area.

Background Information: Amphibalanus amphitrite is hermaphroditic but capable of cross fertilization (Fofonoff et al., 2003). Based on its temperature and salinity tolerances, *A. amphitrite* may be able to reproduce successfully during the warmest period of the year (>10C). Zvyagintsev and Ke (2003) observed that larvae of the barnacle were no longer observed in the water column when water temperatures dropped below 12°C in Peter the Great Bay, Sea of Japan, Russia. *Balanus amphitrite* with developed ovaries, cultured at 10°C ceased breeding, while specimens cultured at 20 °C revealed high breeding activity; similarly barnacles reared at room temperature showed no breeding activity during the winter, when the temperature was lower than 10°C, but commenced breeding when the temperature rose to over 10 °C (El-Komi and Kajihara, 1991). Larvae in the laboratory can complete development at 15°C but at this temperature development is slow and survival can be low (Anil et al., 1995, Qiu and Qian, 1999).

Ice scour and low surface salinity within and downstream from the large estuaries are likely to limit use of habitats from the splash zone to 6 or 7 m depth (Kuzyk et al., (2008). Surface waters may warm sufficiently over the summer to permit reproduction but in winter the water is too cold for these barnacles to survive and establish self-sustaining populations. Further study is needed to clarify the species' tolerances for low temperatures, in particular those of genetic Clade 1 (see Chen et al., 2014). Lack of shelter may also be a limiting factor on the southern coast of Hudson Bay.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Amphibalanus amphitrite.* Information on predators and parasites is limited, and mainly from another barnacle species of the same genus, *A. improvisus*.

Background Information: For the closely related *A. improvisus,* the larvae can be consumed passively by bivalve molluscs and other suspension feeders; adults can be consumed by benthophagous fish, sea stars, flatworms, shrimps and crabs (https://www.cabi.org/isc/datasheet/91903). Few species that prey upon these barnacles have been reported from the RA area. However, *Mytilus trossulus,* a close relative of the blue mussel (*Mytilus edulis*) which can filter balanid larvae, is present (K. Howland, personal communication). The two species are difficult to tell apart and genetic studies suggest that blue mussels reported from the RA area in the past may be *M. trossulus.* In the Baltic Sea the starfish *Asterias rubens* and crab *Carcinus maenas* prey on both the blue mussel and its epibionts, which include *Amphibalanus improvisus* (Laudien and Wahl 1999). Neither predator has been reported from the RA area although a closely related starfish, *Asteria vulgaris,* is present as are crabs of genera other than *Carcinus* (Stewart and Lockhart 2005). Flatworms from the Genus *Stylochus* prey upon *A. improvisus,* including *S. ellipticus* which is a dominant predator of the species on the east coast of North America (Branscomb, 1976), but this flatworm species has not been reported from the RA area. Li et al. (2020) have suggested that predation by gastropods on *A. amphitrite* may be slightly impacted by ocean acidification*.*

From CABI: "Two parasitic crustaceans have been found on *A. improvisus*: the rhizocephalan crustacean *Boschmaella balani* (Bocquet-Védrine, 1972) and the isopod crustacean *Hemioniscus balani* (Tarasov and Zevina, 1957)." https://www.cabi.org/isc/datasheet/91903

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* has the capacity for a wide range of natural dispersal in the RA area*.* There is good evidence for wide range dispersal of the species, which has a protracted planktonic larval stage.

Background Information: Barnacles have a long-lived planktonic larval stage that can remain in the water column for up to two months (Anil et al., 1995). Low temperatures (15°C) and salinities (~10 PSU) reduce larval survival and slow larval development, which can take 18 d at 15°C compared to 4 d at 30°C and 8 d at 10 PSU compared to 5 d at 35 PSU (Anil et al., 1995, Qiu and Qian, 1999). This trait would extend the dispersal period for planktonic larvae in the assessment area. Over time, alongshore currents could disperse them counter clockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait (Granskog et al., 2011).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence of wide range dispersal of *A. amphitrite* via uptake in ballast and/or biofouling of vessels moving within the RA area.

Background Information: Key vectors of anthropogenic dispersal of balanomorph barnacles include transport in ship's ballast water and on fouled hulls, and transport of organisms for aquaculture (Carlton, 2001). Ships visiting Churchill frequently carry live *A. amphitrite* on their hulls (5 of 13 ships tested; averaging 112,859 ± 69,678 individuals/ship) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ±300,133 ind/ship (Chan et al., 2015). Hull fouling and ballast water transported by domestic coastal resupply and other boat traffic could spread barnacles from Churchill north and west to coastal communities around western Hudson Bay, into Baker Lake, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Human transport of barnacles attached to large molluscs or crustaceans is possible but unlikely.

Q9- Impact on population (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus amphitrite* would have low or no impact on population growth in the RA area. There may be limited competition for space with native barnacles and fouling taxa. However, documented impacts on populations have generally been low and others are speculative.

Background Information: In Florida, introduced *A. amphitrite* and native *Amphibalanus eburneus* competed with the Eastern Oyster (*Crassostrea virginica*) for settlement sites, and also affected survival and growth of oysters by settling on their shells (Boudreaux et al., 2009). No differences were observed in impacts to the oysters from the native and non-native barnacle species. Vertical zonation of barnacles and other rocky shoreline taxa probably moderates competition somewhat. *A. amphitrite* affected composition of the Tampa Bay, FL fouling community by creating additional structure for the recruitment and colonization of motile species (Bros, 1987). *Amphibalanus amphitrite* may prevent recruitment of corals by competing for space in marginal environments (Chui and Ang, 2010).Whether establishment of *A. amphitrite* would have the same effects on native invertebrate communities in Hudson Bay is unknown.

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus amphitrite* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There could be competition with native species, and creation of habitat structure may increase diversity.

Background Information: Amphibalanus amphitrite might compete with native barnacles or shellfish for food and space. *Semibalanus balanoides* (formerly *Balanus balanoides*), *Balanus balanus*, and *Balanus crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). *Amphibalanus amphitrite* affected composition of the Tampa Bay, FL fouling community by creating additional structure for the recruitment and colonization of motile species (Bros, 1987). The species may prevent recruitment of corals by competing for space in marginal environments (Chui and Ang, 2010). Whether establishment of *A. amphitrite* would have the same effects on native invertebrate communities in Hudson Bay is unknown.

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species is well documented to create habitat structure.

Background Information: Amphibalanus amphitrite affected composition of the Tampa Bay fouling community by creating additional structure on natural and anthropogenic substrates for the recruitment and colonization of motile species (Bros, 1987). It competes for settling space with other taxa such as barnacles and oysters (Boudreaux et al., 2009).

Q12- Impact on ecosystem function (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus amphitrite* would have low or no impact on ecosystem function in the RA area, as any effects would likely be very localized.

Background Information: In the Indian River Lagoon, FL, introduced *A. amphitrite* and native *A. eburneus* competed with the Eastern Oyster (*Crassostrea virginica*) for settlement sites, and also affected survival and growth of oysters by settling on their shells (Boudreaux et al., 2009). No differences were observed in impacts to the oysters from the native and non-native barnacle species. Whether the same would occur with native barnacle and mollusc species in Hudson Bay is unknown. An assessment of *A. amphitrite* on ecosystem function in the Bering Sea reported no impacts (Alaska Center for Conservation Science, 2017).

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus amphitrite* would have low or no impact associated with diseases, parasites or travelers in the RA area. Effects are likely very localized, if any.

Background Information: The parasitic isopod crustacean *Hemioniscus balani* can affect many North Atlantic barnacle species and if introduced to the RA area, and not already present, could impair or prevent egg production by native barnacles such as *Semibalanus balanoides* and *Balanus balanus,* but *B. crenatus* may not be susceptible (Crisp, 1968). The distribution of this parasite seems to be limited to the Gulf of St Lawrence in the north (Crisp, 1968), so it may not be able to survive in the colder waters of the RA area.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* would have low or no genetic impact on other species in the RA area. No related species of the same genus have been found in the RA area.

Background Information: Semibalanus balanoides (formerly *Balanus balanoides*), *Balanus balanus*, and *B. crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). The distribution of *A. amphitrite* overlaps that of *B. crenatus* elsewhere (e.g., Kerckhof (2002). No reports of hybrids were found but the potential may exist.

Amphibalanus amphitrite has 3 distinct genetic Clades that suggest past population isolation (Chen et al., 2014). Clade 1 is widely distributed in temperate and tropical waters (introduced to Atlantic and Pacific coasts of N. America); Clade 2 occurs in Singapore and has been introduced on the Atlantic coast (North Carolina); Clade 3 is widely distributed in tropical waters (Asia, Middle East, Australia). Human-mediated dispersal may have contributed to present clade overlaps. Environmental tolerances may differ among the clades.

Tsang et al.(2008) have hypothesized that poleward movement of barnacle species might lead to a breakdown of the boundary between species.

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* would have low or no impact on at-risk or depleted species in the RA area. There is little habitat overlap with species at risk in the RA area and likely little effect.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. None of the fish, bird or mammal species at risk are likely to be affected.

Q16- Impact on fisheries (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus amphitrite* would have low or no impact on fished species in the RA area. Overlap between introduced barnacles and harvested species is possible but impacts would likely be limited based on known population/community effects.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Commercially harvested anadromous Arctic char (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus amphitrite* is known to be invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Amphibalanus amphitrite is widely reported as invasive, and a common fouling organism in warm and temperate coastal waters and estuaries worldwide (Fofonoff et al., 2003, Carlton et al., 2011, Chen et al., 2014). The species has regularly settled in Long Island Sound in small numbers but its winter survival there is unknown (Carlton et al., 2011). Its range is expected to expand northward in response to climatic warming (Chan et al., 2015), and tolerance data in Fofonoff et al. (2003) suggest that *A. amphitrite* larvae released with ballast water at Churchill might survive.

Amphibalanus eburneus (Gould, 1841)

Phylum: Arthropoda
  Class: Henauplia
    Order: Sessilia
      Family: Balanidae

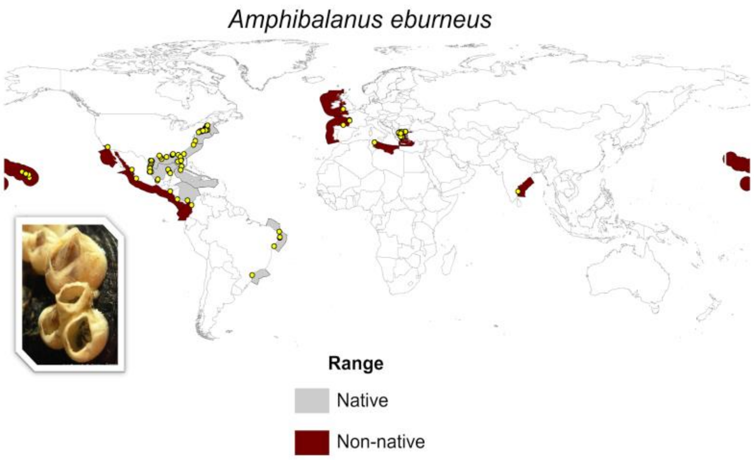

Figure 6: Ecoregions where Amphibalanus eburneus is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Amphibalanus eburneus occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.uuxcu1, 27 June 2017), and NEMESIS (http://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=89621). Picture of A. eburneus modified from https://invasions.si.edu/nemesis/species_summary/89621

CMIST scores for *A. eburneus:*

Mean adjusted Likelihood of Invasion: 2.14

Mean adjusted Impact of Invasion: 1.80

Mean adjusted Overall CMIST score: 3.85

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus eburneus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016) but larvae are being introduced at Churchill in ballast water, and visiting ship's hulls are fouled with adults (Chan et al., 2015). No reports were found of *A. eburneus* in Atlantic Canada; they were found for the first time in BC via subtidal monitoring (establishment unknown) (Therriault et al., 2011).

Q2- Rate of introduction (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* could arrive frequently in high numbers in the RA area. The species is present in connected ports and there is clear evidence of arrival from multiple studies and vectors of transport.

Background Information: Ships visiting Churchill can carry live *A. eburneus* on their hulls (1 of 13 ships tested; 530 individuals/ship) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ±300,133 ind/ship (Chan et al., 2015). Exposure occurs during the open water shipping season. *A. eburneus* is unlikely to be transferred via this route with live commercial shellfish.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Amphibalanus eburneus.* There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support habitat classification.

Background Information: Adult *A. eburneus* occur at depths from the low tidal mark to about 37 m (Voss, 1980). Where they attach to a wide range of hard surfaces including docks, ship's hulls, logs, rocks, and bivalve shells (Henry and McLaughlin, 1975, Fofonoff et al., 2003). These barnacles can live in polluted areas, have a wide salinity tolerance, and prefer low surf areas (Zaitsev and Ozturk, 2001). They are intolerant of currents and more likely to occur on the bottom and in stagnant areas than *A. improvisus* (Zevina and Kuznetsova, 1965).

Larvae in the Black Sea are distributed primarily in the upper 10 m of the water column, within 10 km of the coast (Zaitsev and Ozturk, 2001).

Q4- Survival (suitable environment) (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest the RA area offers a negligible proportion of suitable environmental conditions for *Amphibalanus eburneus.* The species' requirements are not well described and information in the literature conflicts with its apparent distribution.

Background Information: These barnacles are mesohaline to euryhaline. The lower salinity limit for *A. eburneus* seems to be ~5 PSU, below which cyprids in lab experiments do not settle (range tested 2-35 ppt); highest settlement occurred between 15 and 20 ppt (Dineen Jr and Hines, 1994). Successful development was observed at 40 PSU, the highest salinity tested (Bacon, 1976).

The low temperature tolerance of -2°C suggested by Fofonoff et al. (2003), based on the species' occurrence in estuaries with ice cover, as cited in Henry and McLaughlin (1975), is unlikely at the latitude of these estuaries. Further work is needed to establish defensible low-temperature tolerance limits for *A. eburneus*.

Adult *A. eburneus* may survive for a long time in their shell ("latent phase") in the air, or in freshwater (Alexandrov, 1988, Zaitsev and Ozturk, 2001).

Q5- Establishment (reproductive requirements) (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are almost never available for *Amphibalanus eburneus* in the RA area. Temperature tolerances for successful reproduction are uncertain but based on the species' present distribution successful reproduction and/or winter survival at any depth are unlikely.

Background Information: Amphitrite eburneus is hermaphroditic but capable of cross fertilization (Fofonoff et al., 2003). Fofonoff et al. (2003) suggest that the species requires a minimum temperature of 20°C for successful reproduction, based on studies by Scheltema and Williams (1982). However, Scheltema and Williams (1982) did not test colder temperatures, so successful reproduction may occur at significantly lower temperatures. Anil (1991) and Anil et al. (2012) demonstrated experimentally that *A. eburneus* in Japan had successful development across temperatures ranging from 13-30°C and salinities from 10-30 PSU. In the Black Sea, the total duration of *A. eburneus* larval development was 9 days at 22°C and 48 days at 10°C (Alexandrov, 1988, Zaitsev and Ozturk, 2001). The lower salinity limit for *A. eburneus* seems to be ~5 PSU, below which cyprids in lab experiments do not settle (range tested 2-35 ppt); highest settlement occurred between 15 and 20 ppt (Dineen Jr and Hines, 1994). Successful development was observed at 40 PSU, the highest salinity tested (Bacon, 1976). Ice scour is likely to limit use of habitats from the splash zone to 6 or 7 m depth. Lack of shelter may also be a limiting factor on the southern coast of Hudson Bay. Further study is needed to clarify the species' tolerances for low temperatures.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Amphibalanus eburneus*. There is limited information on predators and parasites, mostly from another barnacle species (*A. improvisus*) of the same genus.

Background Information: Amphibalanus eburneus is a general filter-feeder that other barnacle and filter feeders will compete with for food and space (Fofonoff et al., 2003). For the closely related *A. improvisus*, the larvae can be consumed passively by bivalve molluscs and other suspension feeders; adults can be consumed by benthophagous fish, sea stars, flatworms, shrimps and crabs (https://www.cabi.org/isc/datasheet/91903). Few species that prey upon these barnacles have been reported from the RA area. However, *Mytilus trossulus*, a close relative of the blue mussel (*Mytilus edulis*) which can filter balanid larvae, is present (K. Howland, personal communication). The two species are difficult to tell apart and genetic studies suggest that blue mussels reported from the RA area in the past may be *M. trossulus*. In the Baltic Sea the starfish *Asterias rubens* and crab *Carcinus maenas* prey on both the blue mussel and its epibionts, which include *Amphibalanus improvisus* (Laudien and Wahl 1999). Neither predator has been reported from the RA area although a closely related starfish, *Asteria vulgaris*, is present as are crabs of genera other than *Carcinus* (Stewart and Lockhart 2005). Flatworms from the Genus *Stylochus* prey upon *A. improvisus*, including *S. ellipticus* which is a dominant predator of the species on the east coast of North America (Branscomb, 1976), but this flatworm species has not been reported from the RA area.

From CABI: "Two parasitic crustaceans have been found on *A. improvisus*: the rhizocephalan crustacean *Boschmaella balani* (Bocquet-Védrine, 1972) and the isopod crustacean *Hemioniscus balani* (Tarasov and Zevina, 1957)." (https://www.cabi.org/isc/datasheet/91903)

Li et al. (2020) have suggested that predation by gastropods on the closely related *A. amphitrite* may be slightly impacted by ocean acidification.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for wide range dispersal of the species, which has a protracted planktonic larval stage.

Background Information: Low temperatures can slow larval development. In the Black Sea, the total duration of *A. eburneus* larval development was 9 days at 22°C and 48 days at 10°C (Alexandrov, 1988, Zaitsev and Ozturk, 2001). Low summer temperatures in Hudson Bay would extend the dispersal period for planktonic larvae in the assessment area. Over time, alongshore currents could disperse larvae counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait (Granskog et al., 2011).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence for wide range dispersal through uptake in ballast and/or biofouling on vessels moving within the RA area.

Background Information: Key vectors of anthropogenic dispersal of balanomorph barnacles include transport in ship's ballast water and on fouled hulls, and transport of organisms for aquaculture (Carlton, 2001). Ships visiting Churchill can carry live *A. eburneus* attached to their hulls (1 of 13 ships tested; 530 individuals/ship) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ±300,133 ind/ship (Chan et al., 2015).

Hull fouling and ballast water transported by domestic coastal resupply and other boat traffic could spread barnacles from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Human transport of barnacles attached to large molluscs or crustaceans is possible but unlikely.

Q9- Impact on population (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus eburneus* would have high impact in few areas or moderate impact in many areas on population growth in the RA area.

Background Information: In the warm waters of the Caspian Sea, competition for space and food by a mass development of *A. eburneus* considerably reduced the biomass of native species of molluscs and hydroids (Zaitsev and Ozturk, 2001). Under conditions of food shortage *A. eburneus* can prevent the development of other fouling species. Competition by the larvae can be intense as, in some areas, they can constitute up to 90% of the plankton. Whether the same would occur with native barnacle and mollusc species in the RA area is unknown. *Semibalanus balanoides* (formerly *Balanus balanoides*), *B. balanus*, and *B. crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014).

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus eburneus* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There could be competition with native species, and creation of habitat structure may increase diversity.

Background Information: In the warm waters of the Caspian Sea, competition for space and food by a mass development of *A. eburneus* considerably reduced the biomass of native species of molluscs and hydroids (Zaitsev and Ozturk, 2001). Under conditions of food shortage *A. eburneus* can prevent the development of other fouling species. Competition by the larvae can be intense as, in some areas, they can constitute up to 90% of the plankton. Whether the same would occur with native barnacle and mollusc species in the colder environment of Hudson Bay is unknown.

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species is well documented to create habitat structure.

Background Information: Massive colonization by this species may alter benthic/intertidal habitats by the addition of biogenic structure. In some areas, multi-layer foulings of *A. eburneus* can reach 6 cm in thickness (Zaitsev and Ozturk, 2001). It competes for settling space with other taxa such as barnacles and oysters (Boudreaux et al., 2009). Colonies of *A. eburneus* serve as substrates for various species of algae, hydroids, bryozoans and molluscs (Zaitsev and Ozturk, 2001).

Q12- Impact on ecosystem function (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus eburneus* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area.

Background Information: In the warm waters of the Caspian Sea, competition for space and food by a mass development of *A. eburneus* considerably reduced the biomass of native species of molluscs and hydroids (Zaitsev and Ozturk, 2001). Under conditions of food shortage *A. eburneus* can prevent the development of other fouling species. Competition by the larvae can be intense as, in some areas, they can constitute up to 90% of the plankton. Whether the same would occur with native barnacle and mollusc species in the colder environment of Hudson Bay is unknown.

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Amphibalanus eburneus* would have low or no impact associated with diseases, parasites or travelers in the RA area.

Background Information: Viruses of the *Iridovirus* group of the family Iridoviridae have been described from *Amphibalanus eburneus* (Leibovitz and Koulish, 1989). The disease is the first reported viral infection of barnacles or of any other member of the class Cirripedia. The viral agent has a predilection for parenchymatous cells of the barnacle, invading and replicating within these cells. Hypertrophy, degeneration, and necrosis of the parenchymatous cells and tissue resulted from the infection. The virus' ability to survive and infect barnacles indigenous to the RA is unknown.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* would have low or no genetic impact on other species in the RA area. No related species of the same genus have been found in the RA area

Background Information: Semibalanus balanoides (formerly *Balanus balanoides*), *B. balanus*, and *B. crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). No reports of hybrids were found.

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* would have low or no impact on at-risk or depleted species in the RA area.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. None of the fish, bird or mammal species at risk are likely to be affected.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus eburneus* would have high impact in few areas or moderate impact in many areas on fished species in the RA area.

Background Information: Overlap between barnacles and harvested species (bivalves) is possible but impacts would likely be limited based on known population/community effects including directly on mussels, and on plankton. Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Commercially harvested anadromous Arctic charr (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus eburneus* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Amphibalanus eburneus is native to the western Atlantic from the southern Gulf of Maine to Venezuela, and may be cryptogenic from there south to Argentina

(Fofonoff et al., 2003, Carlton et al., 2011). Introduced around the world by shipping, the species has invaded the northeastern Atlantic, Indian, and northwestern and northeastern Pacific oceans. It was also introduced to the Netherlands and southwestern Indian Ocean but does not appear to have become established (Wolff, 2005). Chan et al. (2015), based on tolerance data in Fofonoff et al. (2003), suggest that *A. eburneus* larvae released with ballast water at Churchill might survive. Zevina and Gorin (1975 cited in Zvyagintsev et al. 2011) described the probability of this species colonizing Peter the Great Bay in the Sea of Japan, which is warmer than southern Hudson Bay, as very low, except possibly in shallow, well heated bays.

Amphibalanus improvisus **(Darwin, 1854)**

Phylum: Arthropoda

Class: Hexanauplia

Order: Sessilia

Family: Balanidae

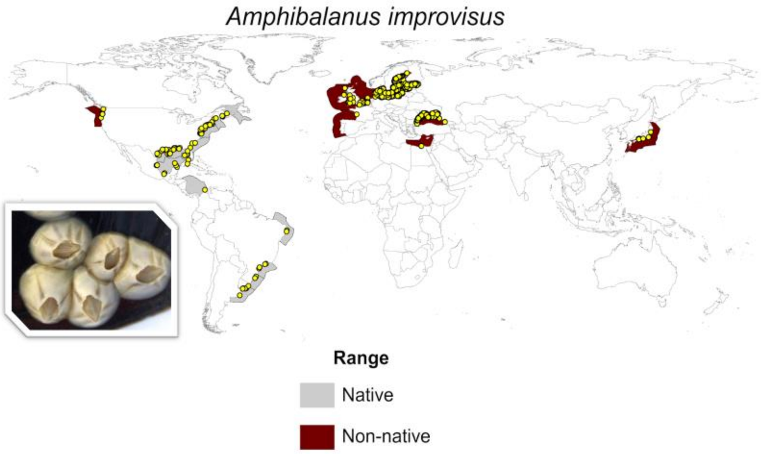

Figure 7: Ecoregions where Amphibalanus improvisus is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Amphibalanus improvisus occurrence points were obtained from OBIS (https://obis.org/). Picture of A. improvisus modified from https://www.cabi.org/isc/datasheet/91903#toPictures.

CMIST scores for *A. improvisus:*

Mean adjusted Likelihood of Invasion: 2.26

Mean adjusted Impact of Invasion: 1.64

Mean adjusted Overall CMIST score: 3.72

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus improvisus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016) but larvae are being introduced in Churchill through ballast water, and visiting ships hulls are fouled with adults (Chan et al., 2015).

Q2- Rate of introduction (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* could arrive frequently in high numbers in the RA area. The species is present in connected ports and there is clear evidence of arrival from multiple studies and vectors of transport.

Background Information: Ships visiting Churchill frequently carry live *A. improvisus* adults on their hulls (3 of 13 ships tested; averaging 11,396 ± 3,511 individuals/ship) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ±300,133 ind/ship (Chan et al., 2015). Exposure occurs annually during the open water shipping season. *Amphibalanus improvisus* is unlikely to be transferred via this route with live commercial shellfish.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Amphibalanus improvisus*. There is reliable information about seabed morphological characteristics in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support habitat classification.

Background Information: Amphibalanus improvisus is eurythermal, euryhaline, and tolerant of low oxygen concentrations. It attaches to hard surfaces, including wooden piers, ships hulls, shells of molluscs and large crustaceans, and rocks at depths from the splash zone to 90 m. The species is intolerant of desiccation (Southward, 1957, Fofonoff et al., 2003, Olenin, 2006, Farrapeira and Calado, 2010). Its range includes ice-covered estuaries (Fofonoff et al., 2003).

Q4- Survival (suitable environment) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Amphibalanus improvisus*. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Rhythmic cirral activity of this eurythermal species declined rapidly above 30°C (not detected above 35.5°C), and below 4°C (Southward, 1957), but continued in some individuals down to -2°C, the lowest temperature tested. After 10 minutes at -2°C, 4 of 10 showed a chill coma. *Amphibalanus improvisus* occurs in the Gulf of St. Lawrence (https://www.gbif.org) where water temperatures drop below zero. Niche modelling results showed that projected environmental conditions under global warming will include a small region of the assessment area as suitable habitat (Goldsmit et al., 2018).

This euryhaline (0-40 PSU) species shows optimum activity at 6 – 30 PSU, with maximal larval settlement in the mid-range of these salinities (Foffonoff et al. 2003). It is tolerant of low oxygen concentration; lives at depths from 0 (splash zone) to 90 m; and is intolerant to desiccation. Low temperature (12°C) narrowed the barnacles' salinity tolerance (Nasrolahi et al., 2013). It can survive weeks of exposure to freshwater with larval settlement occurring at salinities from 2 to 35 PSU (Dineen Jr and Hines, 1992). Small juveniles are more vulnerable to predation (Fofonoff et al., 2003).

Barnacles in nearshore environments can experience substantial diurnal fluctuations in pH (Eriander et al., 2015). In the laboratory, both stable (pH 7.7) and fluctuating (7.5 ≤ pH ≤ 7.9) acidification had a strong negative effect on mean shell strength. Fluctuating acidification did not affect mean growth or shell mineralogy, but caused a 20-fold increase in variance of responses compared with stable acidification (Eriander et al., 2015).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Amphibalanus improvisus* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with a maximum sea surface temperature >=10 °C, the lowest tested limit for reproduction/larval development. There is published information on requirements and there are reliable environmental data layers for the RA area to show a match, but there has not been any testing at temperatures below 10 °C, so it is uncertain if reproduction could be completed at colder temperatures.

Background Information: The species is hermaphroditic but capable of cross fertilization (Fofonoff et al., 2003, Olenin, 2006). Time of naupliar release and cyprid settlement vary.

From CABI: "In the Black Sea (Shalaeva and Lisitskaya, 2004) in 2000-2001, naupliar stages started to appear in plankton when water temperature was 10 °C. Nasrolahi et al. (2006) found with increasing salinity, larval size decreased and development time to cyprid larvae increased (8- 25 ppt takes 7 days, above 36 ppt – 9 days) and larval survival was highest at 12 ppt (60%), [and lower at] 36 ppt (14%)." (https://www.cabi.org/isc/datasheet/91903).

Nasrolahi et al. (2013) and Pansch et al. (2013), both demonstrated experimentally that survival and settlement of larvae was reasonably high across a wide range of salinities (5 ppt -30 ppt) and temperatures (12-28°C), but tended to be higher at lower temperatures, especially at the highest salinity. Dineen Jr and Hines (1992) found that larval settlement occurred at salinities from 2 to 35 PSU. Based on its tolerances for life and reproduction in relation to environmental conditions it seems likely that suitable habitats exist or will if climate change scenarios are accurate, particularly in estuaries.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Amphibalanus improvisus*. There are similar taxa in the RA area that could act as predators. Much of the information on predators and parasites are those not on species found in the study area that would allow for extrapolation to the RA area.

Background Information: Amphibalanus improvisus larvae can be consumed passively by bivalve molluscs and other suspension feeders; adults can be consumed by benthophagous fish, sea stars, flatworms, shrimps and crabs (https://www.cabi.org/isc/datasheet/91903). Few species that prey upon these barnacles have been reported from the RA area. However, *Mytilus trossulus*, a close relative of the blue mussel (*Mytilus edulis*) which can filter balanid larvae, is present (K. Howland, personal communication). The two species are difficult to tell apart and genetic studies suggest that blue mussels reported from the RA area in the past may be *M. trossulus*. In the Baltic Sea the starfish *Asterias rubens* and crab *Carcinus maenas* prey on both the blue mussel and its epibionts, which include *Amphibalanus improvisus* (Laudien and Wahl 1999). Neither predator has been reported from the RA area although a closely related starfish, *Asteria vulgaris*, is present as are crabs of genera other than *Carcinus* (Stewart and Lockhart 2005). Flatworms from the Genus *Stylochus* prey upon *A. improvisus*, including *S. ellipticus* which is a dominant predator of the species on the east coast of North America (Branscomb, 1976), but this species has not been reported from the RA area.

From CABI: “Two parasitic crustaceans have been found on *A. improvisus*: the rhizocephalan crustacean *Boschmaella balani* (Bocquet-Védrine, 1972) and the isopod crustacean *Hemioniscus balani* (Tarasov and Zevina, 1957).” (https://www.cabi.org/isc/datasheet/91903).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for wide range dispersal and barnacles, which are well known for their dispersal capabilities due to extended planktonic larval stage.

Background Information: Low temperatures slow larval development, which can take about 24 d at 12°C but only about 12 d at 20°C (Nasrolahi et al., 2012). This would extend the dispersal period in the assessment area. Over time, alongshore currents (Granskog et al., 2011) could disperse larvae counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait. In the Sea of Japan the average annual rate of spread of *A. improvisus* has been estimated at 13.9 km/year (Iwasaki et al., 2004) and in the Baltic Sea, where it is assisted by counterclockwise currents, at a minimum of 30 km/year (Leppäkoski and Olenin 2000).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence for wide range dispersal via uptake in ballast and/or biofouling on vessels moving within the RA area.

Background Information: Key vectors of anthropogenic dispersal of balanomorph barnacles include transport in ship’s ballast water and on fouled hulls, and transport of organisms for aquaculture (Carlton, 2001). Ships visiting Churchill frequently carry live *A. improvisus* adults on their hulls (3 of 13 ships tested; averaging 11,396 ± 3,511 individuals/ship) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ± 300,133 ind/ship (Chan et al., 2015).

Hull fouling and ballast water transported by domestic coastal resupply and other boat traffic could spread barnacles from Churchill north and west to coastal communities around western Hudson Bay, into Baker Lake, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Human transport of barnacles attached to large molluscs or crustaceans is possible but unlikely.

Q9- Impact on population (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus improvisus* would have low or no impact on population growth in the RA area. Competition for space with native barnacles and fouling taxa is possible, but documented impacts to populations have generally been low and others are speculative.

Background Information: Amphibalanus improvisus may compete with native barnacles or shellfish for food and space (Järvekülg, 1979, Leppäkoski and Olenin 2000, Dürr and Wahl, 2004). The native barnacles *Semibalanus balanoides* (formerly *Balanus balanoides*), *Balanus balanus*, and *B. crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). Introduced *A. improvisus* displaced *Balanus crenatus* to become the dominant intertidal barnacle species in Amursky Bay, Sea of Japan (Ovsyannikova, 2008). However, the impact of *A. improvisus* on populations of other species are not necessarily negative, as their empty shells can provide additional structure that facilitates settlement by some species (Leppäkoski and Olenin 2000), and their empty shells provide new microhabitats for others (Leppäkoski, 1999). Further investigation of ecological effects is required, especially on brackish water systems (Leppäkoski and Olenin 2000, Dürr and Wahl, 2004; https://www.cabi.org/isc/datasheet/91903).

From CABI: "Järvekülg (1979) observed competition for attachment places and food between the filter feeders *A. improvisus*, *Mytilus edulis* (sea forms) and *Dreissena polymorpha* (zebra mussel, brackish water form) in the Pernu Bight [Baltic Sea]. Salinity is the main factor on which the result of competition depends: increased salinity is favourable for the two sea forms and not for zebra mussels. Dürr and Wahl (2004) also described *A. improvisus* as a strong competitor for space, but mentioned that it does not have a negative effect on community diversity in the Baltic. The same authors detected that while *A. improvisus* had no significant effect on recruitment of species, a negative synergistic effect of blue mussels and barnacles on species richness and diversity H-1 (Shannon Index) may be significant in the Western Baltic." (https://www.cabi.org/isc/datasheet/91903).

Shellfish are not cultivated in the RA area but natural populations of blue mussel (*Mytilus edulis*), or closely related *M. trossulus* (K. Howland, personal communication), have been harvested by a commercial test fishery in the Belcher Islands and are occasionally harvested for subsistence by Hudson Bay residents (Stewart and Lockhart, 2005). Natural mussel populations are less vulnerable to heavy settlement of *A. improvisus* larvae and subsequent fouling than cultivated mussels.

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus improvisus* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There could be competition with native species, and creation of habitat structure may alter diversity.

Background Information: Amphibalanus improvisus may compete with native barnacles or shellfish for food and space (Järvekülg, 1979, Leppäkoski and Olenin 2000, Dürr and Wahl, 2004). The native barnacles *Semibalanus balanoides* (formerly *Balanus balanoides*), *Balanus balanus*, and *B. crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). Introduced *A. improvisus* displaced *Balanus crenatus* to become the dominant intertidal barnacle species in Amursky Bay, Sea of Japan (Ovsyannikova, 2008). However, the impact of *A. improvisus* on communities and species diversity appear to be limited and not necessarily negative, as they provide additional structure that facilitates settlement by some species (Leppäkoski and Olenin 2000), and their empty shells provide new microhabitats for others (Leppäkoski, 1999)., Further investigation is required, especially on brackish water systems (Leppäkoski and Olenin 2000, Dürr and Wahl, 2004, https://www.cabi.org/isc/datasheet/91903).

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species is well documented to create habitat structure.

Background Information: Habitat effects are common in other areas where *A. improvisus* has been introduced (Fofonoff et al., 2003). Larvae will settle extensively on eelgrass (*Zostera marinus*), brown kelp (e.g., *Laminaria* spp.) and brown algae (*Fucus vesiculosus*) (Olenin, 2006, Ovsyannikova, 2008). These impacted taxa are present and ecologically important in Hudson Bay (Stewart and Lockhart, 2005). The impact of *A. improvisus* on habitats are not necessarily negative as they can provide additional structure that facilitates settlement by some species (Leppäkoski and Olenin 2000), and their empty shells provide new microhabitats for others (Leppäkoski, 1999). Further investigation of habitat effects is needed, especially in brackish waters (Dürr and Wahl, 2004, Leppäkoski and Olenin 2000, Dürr and Wahl, 2004, https://www.cabi.org/isc/datasheet/91903).

From CABI: "*Amphibalanus improvisus* tends to form dense layers on the surface of artificial structures and other substrata, inhibiting water flow, attracting associate fauna and producing organic debris (Leppäkoski, 1999, Weidema, 2000). The increase in biodeposition and mechanical trapping of organic material caused by *A. improvisus* may result in increasing eutrophication of semi-enclosed systems, providing an important source of material to the benthic environment, including the important detritus food chain (Weidema, 2000, Kotta et al., 2006a, Kotta et al., 2006b). This may potentially increase energy flows from the pelagic system to benthos and cause a shift from pelagic production to benthic production." (https://www.cabi.org/isc/datasheet/91903). However, if introduced to the RA area *A. improvisus* would be very unlikely to cause such organic fouling, eutrophication, and/or changes in energy as there are few artificial structures and no shellfish aquaculture operations for them to colonize.

Q12- Impact on ecosystem function (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus improvisus* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. Some literature suggests *A. improvisus* can alter the energy flow.

Background Information: Introduced *A. improvisus* displaced *Balanus crenatus* to become the dominant intertidal barnacle species in Amursky Bay, Sea of Japan (Ovsyannikova, 2008). Its success and abundance can also prevent the settlement of native *B. crenatus* (Skolka and Preda, 2010). Larvae will settle extensively on eelgrass (*Zostera marinus*), brown kelp (e.g., *Laminaria* spp.) and brown algae (*Fucus vesiculosus*) (Olenin, 2006, Ovsyannikova, 2008). These impacted taxa are present and ecologically important in Hudson Bay (Stewart and Lockhart, 2005).

From CABI: "*Amphibalanus improvisus* tends to form dense layers on the surface of artificial structures and other substrata, inhibiting water flow, attracting associate fauna and producing organic debris (Leppäkoski, 1999, Weidema, 2000). The increase in biodeposition and mechanical trapping of organic material caused by *A. improvisus* may result in increasing eutrophication of semi-enclosed systems, providing an important source of material to the benthic environment, including the important detritus food chain (Weidema, 2000, Kotta et al., 2006a, Kotta et al., 2006b). This may potentially increase the energy flows from pelagic systems to benthos and cause a shift from pelagic production to benthic production. (https://www.cabi.org/isc/datasheet/91903). However, if introduced to the RA area *A. improvisus* would be very unlikely to cause such organic fouling, eutrophication, and/or changes in energy as there are few artificial structures and no shellfish aquaculture operations for them to colonize.

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus improvisus* would have low or no impact associated with diseases, parasites or travelers in the RA area.

Background Information: Two parasitic crustaceans have been found on *A. improvisus*: the rhizocephalan crustacean *Boschmaella balani* (Bocquet-Védrine, 1972) (host-specific to the species) and the isopod crustacean *Hemioniscus balani* (Tarasov and Zevina, 1957). *Hemioniscus balani* can affect many North Atlantic barnacle species and if introduced to the RA area, and not already present, could impair or prevent egg production by native barnacles such as *Semibalanus balanoides* and *Balanus balanus* but *B. crenatus* may not be susceptible (Crisp, 1968). The distribution of this parasite seems to be limited to the Gulf of St Lawrence in the north (Crisp, 1968), so it may not be able to survive in the colder waters of the RA area.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* would have low or no genetic impact on other species in the RA area. No reports were found of species from the same genus in the RA area.

Background Information: Semibalanus balanoides (formerly *Balanus balanoides*), *B. balanus*, and *B. crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). The distribution of *A. improvisus* overlaps that of *B. crenatus* elsewhere (e.g., Kerckhof (2002); Ovsyannikova (2008)). No reports of hybrids were found but the potential may exist (Wrange et al., 2016).

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* would have low or no impact on at-risk or depleted species in the RA area. There is limited habitat overlap with species at risk in the RA area and likely little effect.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay and none of the fish, bird or mammal species at risk are likely to be affected if *A. improvisus* were introduced.

Q16- Impact on fisheries (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Amphibalanus improvisus* would have low or no impact on fished species in the RA area. Overlap between barnacles and harvested species (bivalves) is possible but impacts would likely be limited based on known population/community effects.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) have been harvested by a commercial test fishery in the Belcher Islands and are occasionally harvested for subsistence by Hudson Bay residents (Stewart and Lockhart, 2005). Natural mussel populations are less vulnerable to heavy settlement of *A. improvisus* larvae and subsequent fouling than cultivated mussels. Commercially harvested anadromous Arctic charr (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Amphibalanus improvisus* is invasive somewhere else in the world. Solid evidence of establishment and impacts in invaded areas.

Background Information: Amphibalanus improvisus has a cosmopolitan, cold temperate to tropical distribution (Fofonoff et al., 2003, Olenin, 2006). It can block water intake pipes of factories and power stations. DAISIE (Delivering Alien Invasive Species In Europe) considers it one of the 100 most invasive aquatic marine animals (Olenin, 2006; see also Carlton et al., 2011). Chan et al. (2015), based on tolerance data in Fofonoff et al. (2003), suggest that *A. improvisus* larvae released with ballast water at Churchill might survive. Ware et al. (2016) considered it at risk of introduction into a warming Arctic but may have underestimated the warming required for the species to establish a self-sustaining population.

Austrominius modestus (Darwin, 1854)

Phylum: Arthropoda

Class: Hexanauplia

Order: Sessilia

Family: Austrobalanidae

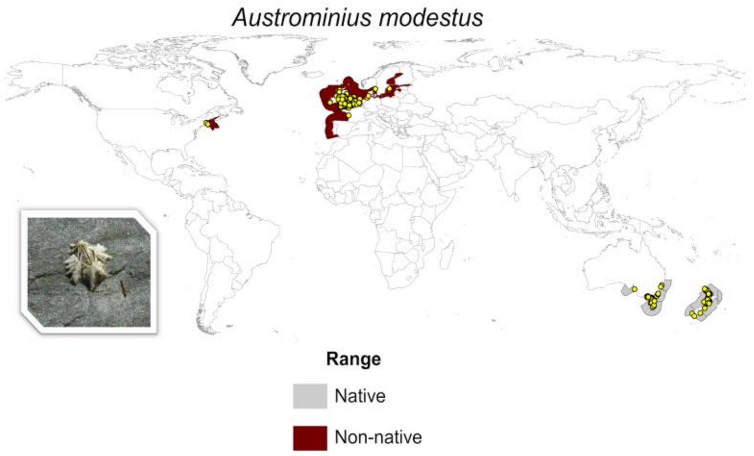

Figure 8: Ecoregions where Austrominius modestus is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Austrominius modestus occurrence points were obtained from OBIS (https://obis.org/). Picture of A. modestus modified from https://www.cabi.org/isc/datasheet/109096

CMIST scores for *A. modestus:*

Mean adjusted Likelihood of Invasion: 2.18

Mean adjusted Impact of Invasion: 1.65

Mean adjusted Overall CMIST score: 3.60

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Austrominius modestus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found of *Austrominius modestus* (formerly *Elminius modestus*) present in Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). However, larvae of this barnacle are being introduced in ballast water and adults were found on the hull of a visiting ship (Chan et al., 2015).

Q2- Rate of introduction (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* could arrive frequently in high numbers in the RA area. The species is present in connected ports and there is clear evidence of arrival from multiple studies and vectors of transport.

Background Information: Ships visiting Churchill can carry live *A. modestus* on their hulls (1 of 13 ships tested; carried 2,005 individuals) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ± 300,133 ind/ship (Chan et al., 2015). Exposure occurs during the openwater shipping season. *Austrominius modestus* is unlikely to be transferred via this route with live commercial shellfish. *Austrominius modestus* is also present in ports that are connected to Deception Bay by shipping (Chan et al., 2012).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Austrominius modestus*. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the habitat classification.

Background Information: Austrominius modestus is particularly common on sheltered shores and in estuaries but can survive on more exposed shores and occurs in the intertidal and shallow subtidal zones to a depth of about 5 m (Crisp, 1958, O'Riordan et al., 2009; and https://www.cabi.org/isc/datasheet/109096). It is eurythermal, euryhaline, resistant to desiccation, tolerant to low oxygen concentrations, and attaches to hard surfaces, including other barnacles, glass, plexiglass, wooden piers, ships hulls, shells of molluscs, macroalgae, tunicates, crabs, and rocks (see also Den Hartog, 1953).

Q4- Survival (suitable environment) (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests the RA area offers a negligible proportion of environmental conditions suitable for *Austrominius modestus*. The species is known to die out when waters freeze over in European locations such as Wadden Sea and Helgolander which are much warmer than the RA area. There is good but somewhat conflicting evidence regarding how low the species' temperature tolerance is.

Background Information: Low water temperatures are likely to restrict the northward spread of this species (Eno et al., 1997). In the North Sea, cold winters (<-20°C air temperature) significantly reduced the adult intertidal and subtidal populations of *A. modestus* resulting in low settling intensity and success the following summer (Harms and Anger, 1989). The associated water temperatures will be substantially higher in Helgolander than those along the Hudson Bay coast (3 °C cf -2 °C). *Austrominius modestus* has not been able to establish in the Danish Wadden Sea, where its population dies out in cold winters and repopulates in mild winters, (Jensen and Knudsen, 2005).

In the laboratory cirral beating of *A. modestus* from the UK ceased below 2 °C (Southward, 1955). This is colder than the species experiences in its native habitat and suggests these invaders have acclimated to the colder waters. *Austrominius modestus* has a lower median lethal air temperature of ca. -5 to -6.6 °C, and a low tolerance to freezing, with water temperatures at a minimum of 4 °C in native range, but can survive at sub-zero temperatures in its invaded European range (O'Riordan et al., 2009; and references within https://www.cabi.org/isc/datasheet/109096). Further evaluation is needed of the species' low temperature tolerances. In Hudson Bay ice scour and cold temperatures are likely to limit use of habitats from the splash zone to 6 or 7 m depth (Kuzyk et al., 2008).

Austrominius modestus displays its greatest activity, measured as cirral and valve movement, when submerged in salinity concentrations close to that of the sea ~33.5‰ (Foster, 1970, Davenport, 1976), and stops all activity outside of the range 17‰ – 53‰ (Foster, 1970).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Austrominius modestus* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with a maximum sea surface temperature >=6 °C, the lowest tested limit for reproduction/larval development. There is published information on the species' thermal requirements and there are reliable environmental data layers for the RA area to show a match, but there has not been any testing at temperatures below 6 °C, so it is uncertain if reproduction could be completed at lower temperatures.

Background Information: This species is an hermaphroditic obligate cross-fertiliser (O'Riordan et al., 2009; CABI and references therein https://www.cabi.org/isc/datasheet/109096). It can produce multiple broods in a year, depending on temperature and food availability. In its native and introduced range, where there is sufficient food and water, and the temperature is above 6 °C, the breeding season appears to be continuous, with occasional peaks. Temperature changes (rise or fall) can induce larval release (Cawthorne and Davenport, 1980). In the North Sea *A. modestus* bred between 7 and 18 °C from May-October, but mostly in July-September (Harms, 1984; cited in O'Riordan et al., 2009). Patel and Crisp (1960) demonstrated experimentally that *A. modestus* could reproduce at 8-9 °C with optimum reproduction at 22-25 °C. The larvae of *A. modestus* from Helgoland developed successfully at a wide range of temperature (6 to 24 °C = range tested) and salinity (20 to 50 ppt; range tested 10 to 50 ppt) (Harms, 1986). Mortality was highest at 10 ppt; only at 12 and 18 °C did a small percentage develop to the cypris. At 30 ppt larval development from nauplii II to cypris takes over 60 days at 6 °C but only 7 days at 24 °C. Cypris larvae are attracted to settle in areas where adults of the species are present (O'Riordan et al., 2009; CABI and references therein https://www.cabi.org/isc/datasheet/109096). Based on its tolerances for life and reproduction in relation to environmental conditions it seems likely that suitable habitats exist in the RA area.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Austrominius modestus*. There is limited information on species-specific controls, mostly general principles, and known controlling species have not been reported from the RA area.

Background Information: In other regions *A. modestus* is eaten by the dog whelk (*Nucella lapillus*) and sea slug (*Onchidoris* sp.), and may be overgrown and smothered by mussels (*Xenostrobus pulex*), oysters (*Crassostrea glomerata*), algae (*Corallina officinalis*), and tunicates (*Botryllus schlosseri*) (O'Riordan et al., 2009; CABI and references therein https://www.cabi.org/isc/datasheet/109096). None of these taxa have been reported from the HB LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014), and it is not known whether other taxa might limit the population growth of *A. modestus* if it becomes established.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for wide range dispersal and the species has good dispersal capabilities due to its extended planktonic larval stage.

Background Information: Larvae of *A. modestus* from Helgoland developed successfully at a wide range of temperatures (6 to 24 °C) and salinities (~20 to 50 PSU) (Harms, 1986). Total duration of the planktonic larvae is longer at lower temperatures and salinities, which could extend dispersal in Hudson Bay relative to warmer waters. At 30 ppt, larval development from nauplii II to cypris takes over 60 days at 6 °C but only 7 days at 24 °C. Over time, alongshore currents (Granskog et al., 2011) could disperse them counter-clockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait .

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence for wide range dispersal via uptake in ballast and/or biofouling on vessels moving within the RA area.

Background Information: Key vectors of anthropogenic dispersal of balanomorph barnacles include transport in ship's ballast water and on fouled hulls, and transport of organisms for aquaculture (Carlton, 2001). Ships visiting Churchill can carry live *A. modestus* on their hulls (1 of 13 ships tested; carried 2,005 individuals) and/or Cirripedia larvae in their ballast water (9 of 32 ships tested; averaging 632,893 ± 300,133 ind/ship (Chan et al., 2015).

Hull fouling and ballast water transported by domestic coastal resupply and other boat traffic could spread barnacles from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of total) in the region, suggesting that this is a plausible vector for within-region spread. Human transport of these barnacles within the region, attached to large molluscs or crustaceans, is possible but unlikely to be a major vector.

Q9- Impact on population (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Austrominius modestus* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. Population effects due to competition for space and resources is well documented (including for species known to occur in the RA area). However, this may vary with salinity and strength of impacts are varied depending on location so there is some uncertainty as to how they would impact native species within the RA area.

Background Information: In Europe, *A. modestus* competes with native barnacles such as *Semibalanus balanoides* (formerly *Balanus balanoides*) and *B. crenatus* for space and resources (Crisp, 1958, Lawson et al., 2004, Witte et al., 2010, Gallagher et al., 2015). The latter species and *B. crenatus* are native to and widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). Following its introduction to German waters ca.1950 *A. modestus* remained relatively rare until ca.1997, when its mean density on oyster beds began increasing exponentially, a trend that continued annually until at least 2008 (Witte et al., 2010). This increase is correlated with increasing air temperatures over the period 1998-2000, which may have enhanced the species' breeding success allowing it to overtake the native barnacles in abundance. In Lough Hyne, Ireland, *A. modestus* totally dominates the barnacle fauna in areas influenced by freshwater (Lawson et al., 2004). Once established in the North Basin, the sheltered nature of the Lough, combined with high summer temperatures and limited circulation probably fostered retention of larvae and heavy spatfall of *A. modestus*. Farther north, on the Isle of Cumbrae, Scotland, where it was introduced ca.1955, *A. modestus* has not outnumbered native barnacle species (e.g., *S. balanoides*), which continue to recruit at high densities (Gallagher et al., 2015). This recruitment probably reflects the ability of these species to withstand the environment, and could change in response to warming and selection.

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Austrominius modestus* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There is a moderate amount of information to suggest creation of habitat structure may alter diversity, and there could be competition with native species, etc, but impacts have varied depending on local conditions, so there is some uncertainty as to how they would impact native species within the RA area.

Background Information: Austrominius modestus may compete with native barnacles or shellfish for food and space (Crisp, 1958, Lawson et al., 2004, Witte et al., 2010, Gallagher et al., 2015). *Balanus balanus, and Balanus crenatus* are widely present in the Hudson Bay LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014). The outcome of competition appears to be sensitive to a range of factors, particularly water temperature, salinity, depth, and exposure/shelter.

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species is well documented to create habitat structure.

Background Information: Austrominius modestus competes for settling space with other taxa such as barnacles and oysters (O'Riordan et al., 2009; CABI and references within https://www.cabi.org/isc/datasheet/109096) and colonies of *A. modestus* serve as substrates for various species of molluscs, algae, and tunicates.

Q12- Impact on ecosystem function (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Austrominius modestus* would have low or no impact on ecosystem function in the RA area*.* Effects are likely to be very localized.

Background Information: It may outcompete some intertidal species in sheltered areas over a narrow range of depths (e.g., *Balanus crenatus*, *Semibalanus balanoides*, *Mytilus edulis*); when present in high abundances its high reproductive output could alter the composition of the summer zooplankton, presumably at the expense of other taxa (O'Riordan et al., 2009).

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Austrominius modestus* would have low or no impact associated with diseases, parasites or travelers in the RA area*.*

Background Information: Austrominius modestus is host to various European parasite species, usually at low prevalence (O'Riordan et al., 2009; CABI and references within https://www.cabi.org/isc/datasheet/109096). But, subtidal *A. modestus* can have a high prevalence of infection by the parasitic isopod crustacean *Hemioniscus balani*. This parasite can infect many North Atlantic barnacle species and if introduced to the RA area, and not already present, could impair or prevent egg production by native barnacles such as *Semibalanus balanoides* and *Balanus balanus*, but *B. crenatus* may not be susceptible (Crisp, 1968, White, 2008). The distribution of *H. balini* seems to be limited to the Gulf of St Lawrence in the north (Crisp, 1968), so it may not survive in the colder waters of the RA area.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* would have low or no genetic impact on other species in the RA area.

Background Information: No other *Austrominius* spp. are present in the HB LME (Stewart and Lockhart, 2005, Goldsmit et al., 2014) and no reports of hybrids with *A. modestus* were found.

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* would have low or no impact on at-risk or depleted species in the RA area. There is little habitat overlap of *A. modestus* with species at risk in the RA area and likely little effect.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay and none of the fish, bird or mammal species at risk are likely to be affected.

Q16- Impact on fisheries (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Austrominius modestus* would have low or no impact on fished species in the RA area. There is overlap between barnacle and harvested species (bivalves) but impacts would likely be limited based on known population/community effects.

Background Information: Austrominius modestus settles on mussels, periwinkles and oysters, fouling commercial rearing operations (O'Riordan et al., 2009; CABI and references therein https://www.cabi.org/isc/datasheet/109096). Shellfish are not cultivated in the assessment area but natural populations of blue mussel (*Mytilus edulis*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. However, settling by *A. modestus* is unlikely to significantly damage these fisheries. Commercially harvested anadromous Arctic char (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Austrominius modestus* is invasive elsewhere in the world. There is good evidence of the species' establishment and impacts in invaded areas.

Background Information: Austrominius modestus is native to New Zealand and either native or cryptic in Australia (O'Riordan et al., 2009), invasive in Europe, where it was introduced by shipping or flying boats (Eno et al., 1997); and it was introduced but did not establish in South Africa. Crisp (1958) suggested that ships fouled with *A. modestus* and anchored for some time in an enclosed harbour (e.g., Southampton at the start of WWII) might have provided sufficient larvae for establishment to occur in the UK. The barnacle's obligate cross-fertilizing hermaphroditism is a key factor limiting establishment as the sessile adults must be within 5 cm of one another to breed: "critical breeding density" (Crisp, 1958). The success of this barnacle in Europe has been attributed to its being eurythermal and euryhaline, its rapid growth, high reproductive capacity, long settlement period, generalist feeding habits and tolerance of turbid waters (O'Riordan et al., 2009). While *A. modestus* is a temperate water species, individuals in Europe are subject to considerable genetic selection pressure towards cold adaptation (Harms and Anger, 1989). Larvae from Helgoland, Germany, were better adapted to cooler temperatures than those in their native New Zealand (Harms, 1986).

ZOOBENTHOS – CRABS

Carcinus maenas (Linnaeus, 1758)

Phylum: Arthropoda

Class: Malacostraca

Order: Decapoda

Family: Carcinidae

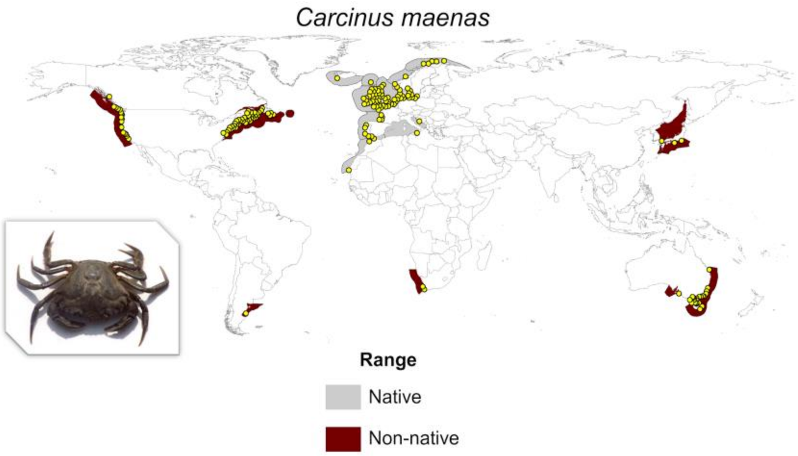

Figure 9: Ecoregions where Carcinus maenas is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Carcinus maenas occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://api.gbif.org/v0.9/occurrence/download/request/0009134-140429114108248.zip, 10 June 2014), NEMESIS (http://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=98734). Picture of C. maenas modified from https://www.cabi.org/isc/datasheet/90475.

CMIST scores for *C. maenas:*

Mean adjusted Likelihood of Invasion: 2.24

Mean adjusted Impact of Invasion: 2.44

Mean adjusted Overall CMIST score: 5.47

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Carcinus maenas* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). *Carcinus maenas* was first detected in Canadian waters in 1951 in the Bay of Fundy (Leim, 1951) and has since spread to NS, NB, PEI (Klassen and Locke, 2007) and NFLD (DFO 2011). It has been present along the west coast of Vancouver Island since 1997/98 (Gillespie et al., 2007).

Q2- Rate of introduction **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Carcinus maenas* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Ballast water is the most likely transport vector for introduction of *C. maenas* to Hudson Bay. Propagule pressure is unknown but *C. maenas* is present at ports directly connected to Churchill and Deception Bay by ships travelling in ballast (Chan et al., 2012). Therriault et al. (2008b) assessed ballast water as the single most important vector for the initial establishment of green crabs, and larval drift as most important for secondary dispersal on the Atlantic and Pacific coasts of Canada.

Strong currents that flow east through southern Hudson Strait and then south along the Labrador coast should limit larval dispersal from Newfoundland into the Hudson Bay LME. Distance from existing *C. maenas* populations will limit transport by small boats and there is no aquaculture in the HB LME at present.

Q3- Survival (suitable habitat) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable habitat conditions for *Carcinus maenas*. There is reliable information about seabed morphological characteristics in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: Postlarval green crabs are commonly found in intertidal and subtidal zones to 5-6 m depth, but occur up to 55 m depth (Crothers, 1968, Grosholz and Ruiz, 1996, Grosholz and Ruiz, 2002, Ray, 2005, Klassen and Locke, 2007). They occur in unstructured sandy and muddy bottoms, are commonly found in salt marshes and seagrass beds, and also use woody debris and rocky substrate. These habitats span a wide range of exposure gradients, from estuaries and sheltered embayments to exposed outer coasts. For reasons unknown, use of rocky habitats appears to be relatively limited in western North America and Tasmania.

Very large areas of southern Hudson Bay, most of James Bay, and the eastern half of Foxe Basin offer habitat shallower than 50 m deep with mud or sand bottoms (Stewart and Lockhart, 2005). The coasts of James bay offer more sheltered embayments and eelgrass beds; both Hudson and James bays have many large estuaries.

Q4- Survival (suitable environment) (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests the RA area offers a negligible proportion of suitable environmental conditions for *Carcinus maenas*. Modelling and experimental data suggest it would not survive well in the RA area. Also, although it is well established and survives winters in the southern Gulf of St. Lawrence, it does not appear to be expanding its range north into colder waters. Although the species requirements are well documented, based on northern limits, they should be able to survive in conditions within a moderate proportion of the RA area. Existing modelling and experimental studies suggest a negligible proportion of suitable habitat within the RA area.

Background Information: Non-larval crabs prefer temperatures of 3° to 26°C (eurythermic) and salinities of 10 to 30 ppt (mesohaline to polyhaline) (Grosholz and Ruiz, 2002). Their critical thermal maximum is 31-36°C, depending upon acclimation conditions (Cuculescu et al., 1998), and platelet ice appears in adult crab tissue at -1.3°C to -1.5°C (Kelley et al., 2013). BC crabs held for 18 weeks at or below 5°C survived and fed but moved slower than in warmer water (Kelley et al., 2013). Growth was suppressed and molting did not occur at <10°C; feeding stops below 2°- 6°C and resumes in spring at about 10°C (Klassen and Locke, 2007). Die-offs of green crabs occur during severe winters with sustained water temperatures ≤0°C (Cohen et al., 1995).

Adults live in salinities of 4 to 34 ppt but tolerate short-term exposure to salinities as low as 1.4 ppt and high as 54 ppt (Cohen et al., 1995). Salinity below 10 ppt elicits an escape response (McGaw et al., 1999), and reduces their ability to adapt to hypoxia (Legeay and Massabuau, 2000). They can survive a week with little available seawater (Darbyson, 2006).

Low temperature tolerance and invasion predictions are often based on remote sensing or modelling of surface temperatures that may not account for sea ice cover. To establish environmental suitability, better data are needed on the low temperature tolerance of *C. maenas* at different salinities, and on the year-round nearshore temperature and salinity profiles. Niche modelling results showed that projected environmental conditions under global warming will include a small region of the assessment area as suitable habitat, while current conditions suggest there is negligible suitable habitat under current conditions (Goldsmit et al., 2018).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Carcinus maenas* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >=6 °C and >=10 °C (the lowest limit at which reproduction reported to occur) resulted in selecting a moderate amount of habitat available in the RA area. Although there is substantial published information on requirements and there are reliable environmental data layers for the RA area to show a match, there appear to be high levels of plasticity within the species and variable findings on abilities to reproduce at temperatures below 10 °C, so it is uncertain if the life cycle could be completed at lower temperatures for more cold-adapted populations that exist in areas such as Newfoundland.

Background Information: Temperatures required for successful reproduction appear to limit green crab distribution. Eggs are brooded externally by the female. Incubation time increases as temperature decreases (19 d at 21.0°C; 38 d at 15°C; 66 d at 11.0°C) (Wear, 1974). In the laboratory, larvae have been successfully reared from hatching through metamorphosis to the juvenile (C1) stage at 10° to 22.5°C (de Rivera et al., 2007); test temperature range: 4-30°C) and at 10°C over salinities from 20 to 35 ppt (Nagaraj, 1993). Temperatures below 10°C were not tested by Wear (1974) or Nagaraj (1993). Larval stages were found off southern NS at temperatures ranging from 5 to 18°C Roff et al. (1984), but it is unknown whether development was occurring in the lower temperature range. However, Best et al. (2017) found that females in Newfoundland (Placentia Bay) were ovigerous in May-June at temperatures of 6-10°C with estimated larval release as early as late May-early June and settlement/first moult occurring by early July. Freshly hatched zoea larvae survived at salinities <15 ppt, but did not fully develop through the life cycle, while metamorphosis to the megalopa stage required salinities ≥20 ppt (Anger et al., 1998). In the laboratory the planktonic larval period is 17–27 days at 25°C and 44–80 days at 12°C (Carlton and Cohen (2003) and references therein). Hudson/James Bay is colder so incubation time and larval development will be slower.

Compared with warmer water native populations in southern Britain and Holland, where the mean water temperature is 4-5°C warmer, *C. maenas* along the Maine coast have later settlement of megalopae, slower growth, delayed maturity, longer generation time, and a longer life span (Berrill, 1982). The time required for each of these life history attributes may increase in the colder water of Hudson Bay.

Q6- Establishment (natural control agents) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests natural control agents are unlikely to affect *Carcinus maenas* population growth in the RA area. Although there is considerable evidence that predation can occur on this species, there is little evidence suggesting this can curtail its population growth. Although there are known parasitic biocontrol agents for crabs, there are none specific to this species and uncertainty regarding native parasites in the RA area. Despite knowing much about the species' biology and interactions, it is unclear the extent to which population growth could be limited by predators, parasites, etc.

Background Information: From Klassen and Locke (2007): "High mortality from predation during settlement and early post-settlement was recorded in all habitats in Sweden by cannibalistic juvenile green crabs and shrimps (brown shrimp *Crangon crangon*, grass shrimp *Palaemon elegans*) with average mortality of 22% and 64% of the settling crabs/3 days, respectively (Moksnes, 2002). The predominant predators of green crabs include fishes ["In North America, they are eaten by striped bass *Morone saxatilis* (Nelson et al., 2003). Cohen et al. (1995) reviewed literature listing, in addition, two sculpins, three gobies, various gadids and flatfish, a ray and a shark as predators of green crab in the Atlantic. Fish preying on green crabs in San Francisco Bay included staghorn sculpin *Leptocottus armatus*, Pacific tomcod *Microgadus proximus*, starry flounder *Platichthys stellatus*, English sole *Parophrys vetulus*, Pacific sanddab *Citharichthys sordidus*, pile perch *Damalicthys vacca*, white surfperch *Phanerodon furcatus*, rubberlip surfperch *Rhacochilus toxotes*, striped bass *Morone saxatilis*, white croaker *Genyonemus lineatus*, white sturgeon *Acipenser transmontanus*, green sturgeon *Acipenser medirostris*, bat ray *Myliobatis californica*, big skate *Raja binoculata*, leopard shark *Triakis semifasciata*, and brown smoothhound shark *Mustelus henlei* (Cohen et al., 1995)], birds ["In North America, sandpipers, sanderling, curlew, the great blue heron *Ardea herodias*, cormorants, ducks including the mallard *Anas platyrhyncha*, and gulls, feed on green crabs" (Cohen et al., 1995). Ellis et al. (2005) found that crabs in the Gulf of Maine were preyed on by Great Black-backed Gulls but were not a preferred food item; green crab was a major prey of herring gull, *Larus argentatus* in the UK (Sibly and McCleery, 1983, Dumas and Witman, 1993)"], and larger decapods [European reports of predation included another by the brown shrimp *Crangon crangon* ((Pihl and Rosenberg, 1984), as well as velvet swimming crabs *Liocarcinus puber* (Rheinallt, 1986)]. Adult rock crabs *Cancer irroratus* preyed on adult green crabs in the laboratory (Elner, 1981). Predation pressure by native rock crabs of *Cancer* spp. may influence habitat preference in green crabs on the Pacific coast (Hunt and Yamada, 2003). The blue crab *Callinectes sapidus* may limit both abundance and geographic range of green crabs on the Atlantic coast (De Rivera et al., 2005). Adult American lobsters *Homarus americanus* in aquaria readily consume green crabs (Elner, 1981; Locke pers. Obs.). Green crabs were a dominant food in the diet of coastal populations of mink *Mustela vison* and otters *Lutra lutra* (Mason and Macdonald, 1980, Dunstone and Birks, 1987). They were also consumed by harbour seal *Phoca vitulina* (Sergeant 1951 cited in Cohen et al. 1995)."

There is good evidence that many animals prey on green crab including several species known to be in the RA area, although there is no indication that this slows their population growth. The green crab's success as an invader suggests that predators, competitors, parasites, diseases, and disturbance are seldom able to slow their population growth. Competition from other crab species, such as *Hemigrapsus sanguineus*—another invader (Jensen et al., 2002, Lohrer and Whitlatch, 2002a, b, Griffen et al., 2008), may be an exception, but whether native crab species in Hudson Bay would provide similar competition is unknown.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* has the capacity for a wide range of natural dispersal in the RA area. There is good supporting evidence for wide range dispersal based on their extended planktonic larval stage and history of spread.

Background Information: Therriault et al. (2008b) assessed larval drift as the most important vector for secondary dispersal of green crab on the Atlantic and Pacific coasts of Canada. In the laboratory the species' planktonic larval period can increase from 17–27 days at 25°C to 44–80 days at 12°C, with its planktonic period potentially extended a few weeks more by drift of post larval crabs (Carlton and Cohen, 2003 and references therein). Larvae that survive after release into the HB LME could have an even longer dispersal period in which to find suitable habitat. Alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

From Klassen and Locke (2007): "Larvae have the potential to disperse over considerable distances given that green crab larval stages must develop in open waters for >50 days, and indeed may remain in the water column for >80 days. Behrens Yamada et al. (2005) attributed dispersal of green crabs along the Pacific coast to larval transport by ocean currents associated with an unusually intense El Niño effect. Northward-moving coastal currents transported larvae up to 50 km/d during the El Niño of 1998 (Behrens Yamada and Becklund, 2004). Oceanographic current changes associated with global climate change are likely to affect the distances and directions of future range expansion (Roman 2006). Dispersal by adults and juveniles is relatively local in nature. There have been no records of adult or juvenile green crabs at sea on floating algae or logs (Cohen et al., 1995). In western Sweden, most green crabs immigrated to coastal embayments as pelagic megalopae, and there was little post-metamorphosis dispersal by juvenile crabs (Moksnes 2002)."

On the west coast of Canada and the USA, green crabs dispersed northward about 1500 km in 12 yr (Jamieson et al., 2002). The strong recruitment event and major range expansion that took place in 1998 was believed to have been the result of unusually strong northward-moving coastal currents of up to 50 km/day, which occurred between November 1997 and February 1998 (Jamieson et al., 2002, Behrens Yamada and Becklund, 2004). In contrast, following its arrival in western North America in 1989, green crab remained limited to San Francisco Bay until 1993, when it spread 80 km northward, and 1994, when it spread 125 km southward (Grosholz and Ruiz, 1996). Mean annual range expansion over the five years of 20 km/yr northward and 31 km/yr southward was close to the mean range expansion for marine species generally (Grosholz and Ruiz, 1996). Northward expansion of green crab from New England to Nova Scotia averaged 63 km/yr, but was very episodic (Grosholz and Ruiz, 1996). Within the southern Gulf of St. Lawrence, episodic range expansions of up to 100 km in a year have been observed (Locke et al. unpub. data). In South Africa, range expansion averaged 16 km/yr from 1983 to 1992 (Grosholz and Ruiz, 1996).

Carcinus maenas is spreading actively along both temperate coasts in Canada at this time (Audet et al. 2008; Brasseale et al. 2019), suggesting it has great potential for fairly long-distance dispersal as larvae.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Carcinus maenas* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal through shipping, via uptake in ballast. Importance of other vectors such as biofouling is less clear and some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: From Klassen and Locke (2007): "The vast majority of green crab invasions throughout the world have been attributed to transport by human agents. Human-mediated dispersal methods include: ballast water, other shipping vectors e.g., seawater pipe systems (sea chests), shipment of commercial shellfish/aquaculture products, bait release, release as a potential food resource, traps and cages, deliberate or accidental release from research/education facilities, marine construction equipment, movement of sediments/sand, and historical vectors such as dry ballast (Cohen et al., 1995, Grosholz and Ruiz, 2002). One of the major vectors for green crab invasions has been shipping (Cohen and Carlton, 1995, Cohen et al., 1995). Carlton and Cohen (2003) documented three major episodes of anthropogenic transport of green crabs to North America: around 1800, the 1850's to 1870's and the 1980's to 1990's. The invasions of the 1800's were largely attributable to transport of adult crabs in dry ballast and ships hulls. Subsequent ballast-mediated invasions would have been in water ballast. Those of the 20th and 21st centuries have been due to a greater variety of transport mechanisms (ships hulls, ballast water, drilling platforms, fishery product transport, scientific research, aquarium releases, etc.). They attributed the observed increase in recorded invasions to a world-wide increase in shipping. Darbyson (2006) suggested that while commercial shipping may have been a factor in the arrival (primary invasion) of green crabs to the southern Gulf of Saint Lawrence, local dispersal (secondary spread) within the Gulf was likely caused by fishing, aquaculture and recreational boating activity. The ability of green crabs to survive for extended periods in the bilges of boats and other apparently unfavorable conditions was well known to fishermen in Maine in the 1950s (Dow and Wallace, 1952). Dow and Wallace report having left green crabs in bags of brackish water in the trunk of a car for over 24 hr, transferring them to fresh water for 6 hr, then dumping out the water and leaving the crabs in the damp bags for a further two days until they finally died. Darbyson (2006) found that green crabs could readily survive 5 days out of water in black fish crates in summer. Green crabs may survive 8 days out of water, although the conditions of the test were not reported (JCG Resource Consultants, 2002)."

Ballast water transported by domestic coastal resupply and other boat traffic, and possibly hull fouling, could spread green crabs from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2015) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of total) in the region, suggesting that this is a plausible vector for within-region spread. With the exception of ship-related movements, the importance of other types of movements (e.g., smaller pleasure and fishing vessels) in HBC is not well known. There is no aquaculture in the region at present where *C. maenas* could be transported with fouled equipment.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* would have high impact in many areas on population growth in the RA area. This is a well-known invader that causes great impacts to a variety of populations and communities.

Background Information: Carcinus maenas eat organisms from at least 104 families and 158 genera in 5 plant and protist and 14 animal phyla (Cohen et al., 1995). They prey on algae, sessile and mobile epifauna, and shallowly buried infauna. Diet varies widely among sites but molluscs and crustaceans are often predominant prey (Grosholz and Ruiz, 1996). Predation by these invasive crabs has significantly reduced many native shellfish populations (Grosholz and Ruiz, 1996, Grosholz and Ruiz, 2002, Fofonoff et al., 2003). The indirect effects of their predation can be trophic cascades.

Carcinus maenas eats clams such as *Mytilus edulis* and *Macoma balthica* (Cohen et al., 1995), and the periwinkle *Littorina saxatilis* (Eastwood et al., 2007) that occur widely in the HB LME. Damaging these populations could adversely affect species that feed upon them year-round such as the Atlantic walrus (*Odobenus rosmarus rosmarus*), bearded seal (*Erignathus barbatus*), and Hudson Bay eider (*Somateria mollissima*). *Carcinus maenas* may compete for space and/or prey with native crabs and shorebirds. Their burrowing for shelter and digging for prey can reduce eelgrass beds (DFO 2011), which provide habitat critical to the ecology of migratory brant geese (*Branta bernicla*) and other species, especially in James Bay (Stewart and Lockhart, 2005). Green crabs are eaten by many species that occur in the HB LME or are represented there by closely related species, e.g. cod (*Gadus* spp.), sculpin (*Cottus* spp.), flounder (*Platichthys* spp.); loons (*Gavia immer*), eiders, herring gulls (*Larus agrentatus*), and seals (e.g. *Phoca vitulina*) (Crothers, 1968). Ice cover and scour may alter impacts of *C. maenas* on communities in the HB LME compared to those occurring in warmer waters.

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* would have high impact in many areas on communities in the RA area. This is a well-known invader that causes great impacts to a variety of populations and communities.

Background Information: Carcinus maenas can fundamentally alter marine communities (Grosholz and Ruiz, 1996, Grosholz and Ruiz, 2002, Fofonoff et al., 2003, Ray, 2005, Klassen and Locke, 2007; DFO 2011). The impacts are quite consistent throughout the crab's native and introduced range. For non-commercial prey species, large effects that often include population declines are attributed to predation by *C. maenas*. For commercial and aquaculture species, similar effects (high mortality rates and reduced yields) are attributed to *C. maenas*, representing significant economic losses. Such strong direct effects are believed to result in many indirect effects on plant, invertebrate, and vertebrate species including ecologically sensitive shorebirds and trophic cascades.

If green crabs are introduced to Hudson Bay they are likely to grow larger in the colder waters than those in warmer waters of the Atlantic and Pacific coasts (Kelley et al., 2015). This may alter the impact of this species relative to predictions based upon attributes in its native range, since larger green crabs can forage to deeper depths in the sediment, and are capable of taking larger prey (Jensen and Jensen, 1985, Grosholz and Ruiz, 1996). Ice cover and ice scour may alter the impacts of *C. maenas* on communities in the HB LME compared to those occurring in warmer waters.

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* would have high impact in many areas on habitat in the RA area. This is a well-known invader that causes great impacts to a variety of populations and communities, including eelgrass and thus impacting habitat directly this way.

Background Information: Carcinus maenas dig through the top few centimeters of sediment, sometimes up to 15 cm deep, searching for prey (Cohen et al., 1995). This bioperturbation alters the natural habitat, particularly where the crabs are abundant and water is shallow, and where beds of bivalves such as soft-shelled clams or scallops are present (DFO 2011). While the bivalve populations are often reduced, other species may benefit from the reduction of predators or of competitors for food or space (Cohen et al., 1995, Grosholz and Ruiz, 2002, Fofonoff et al., 2003).

Green crabs burrowing for shelter and digging for prey can substantially reduce the coverage of ecologically important eelgrass beds (DFO 2011), which provide habitat that is critical to the ecology of migratory waterfowl (e.g., brant geese) and other species, especially in James Bay (Stewart and Lockhart, 2005). Ice cover and ice scour may alter the impacts of *C. maenas* in intertidal and upper subtidal communities in the HB LME compared to what occurs in warmer waters.

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* would have high impact in many areas on ecosystem function in the RA area. This is a well-known invader that causes great impacts to a variety of populations and communities and thus ecosystem functioning.

Background Information: Green crabs are an important predator in shallow waters where they have significant direct and indirect impacts on soft-sediment and rocky shore communities through predation, competition, and bioperturbation (Grosholz and Ruiz, 2002, Fofonoff et al., 2003). They are capable of significantly reducing bivalve, gastropod, and native crab populations by predation; native crab and shorebird species by competition; and eelgrass beds by bioperturbation. Experimental studies have measured significant negative effects of green crab predation on the abundance of gastropod grazers, and the removal of these grazers has significant indirect effects on the community. As a novel and voracious top predator, *C. maenas* can cause trophic cascades, the impacts of their predation and activities extending through several trophic levels and affecting organisms that do not interact directly with the crabs.

Ice cover and ice scour may alter the impacts of *C. maenas* on ecosystem function in the HB LME compared to what occurs in warmer waters.

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Carcinus maenas* would have low or no impact associated with diseases, parasites or travelers in the RA area. The main vector for introduction of *C. maenas* is ballast water, where it is in larval stage. Hence all parasites and diseases that can be transported by adult crabs would be very low. However, there may be a possibility of expansion into the RA area by movement of adults (with associated parasites and diseases) established in adjacent regions (Newfoundland) which would add some uncertainty as to whether risk could be higher.

Background Information: Carcinus maenas is host to a broad range of parasites and pathogens but the prevalence and intensity of infection, and species diversity, are low for introduced populations (Grosholz and Ruiz, 2002). The extent to which these parasites might be new to and/or affect biota in Hudson Bay is unknown. As the main vector of introduction is ballast water carrying the larval stage, all parasites and diseases that can be transported by adult crabs would be very low.

Epifauna that may encrust the integument of *C. maenas* include invasive species such *Molgula manhattensis* and *Botryllus schlosseri* (Crothers, 1968).

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* would have low or no genetic impact on other species in the RA area*.* There are no related species of the same genus in the RA area.

Background Information: There are no native crabs of the same genus in the RA area, so *C. maenas* would be unable to hybridize with local species (Stewart and Lockhart, 2005, Goldsmit et al., 2014) https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Carcinus maenas* would have high impact in few areas or moderate impact in many areas to at-risk or depleted species in the RA area*.* Clear potential for major impacts due to alterations to important eelgrass communities in coastal zones, possibly affecting at risk species; however interactions and potential effects in the RA are not known.

Background Information: Carcinus maenas digs through the top few centimeters of sediment, sometimes up to 15 cm deep, searching for prey (Cohen et al., 1995). This bioperturbation alters the natural habitat, particularly where the crabs are abundant and water is shallow, and where beds of bivalves such as soft-shelled clams or scallops are present (DFO 2011). Green crab burrowing for shelter and digging for prey can substantially reduce the coverage of ecologically important eelgrass beds (DFO 2011), which provide habitat that is critical to the ecology of migratory waterfowl (e.g., brant geese) and other species, especially in James Bay (Stewart and Lockhart, 2005).

There are no invertebrate or plant species at risk in Hudson Bay, however, fish and bird species at risk may be affected through habitat impacts and trophic cascades. Two marine mammal species at risk, Beluga whales (*Delphinapterus leucas*) (COSEWIC 2004) and Atlantic walruses (*Odobenus rosmarus*) (COSEWIC 2006) might benefit from *C. maenas* invasion by preying upon the crabs, although any benefit to walruses may be outweighed by competition with the crabs for mollusc prey.

Q16- Impact on fisheries (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Carcinus maenas* would have high impact in many areas on fished species in the RA area. There is clear potential for major impacts due to effects on eelgrass, associated communities, and *Mya truncata* in coastal zones, possibly affecting both fished/harvested and ecologically important species.

Background Information: Carcinus maenas digs through the top few centimeters of sediment, sometimes up to 15 cm deep, searching for prey (Cohen et al., 1995). This bioperturbation alters the natural habitat, particularly where the crabs are abundant and water is shallow, and where beds of bivalves such as soft-shelled clams or scallops are present (DFO 2011). Green crab burrowing for shelter and digging for prey can substantially reduce the coverage of ecologically important eelgrass beds (DFO 2011), which provide habitat that is critical to the ecology of migratory waterfowl (e.g., brant geese) and other species, especially in James Bay (Stewart and Lockhart, 2005).

Shellfish are not cultivated in the HB LME. Natural populations of blue mussel (*Mytilus edulis*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, and are occasionally harvested for subsistence by Hudson Bay residents (Stewart and Lockhart, 2005). Greenland glass scallop (*Similipecten greenlandicus*) have been harvested by test fisheries in northeastern Hudson Bay. In situ experiments on the New Hampshire coast found *M. edulis* can comprise up to 45% of the *C. maenas* diet (Griffen and Delaney, 2007). Green crabs are important predators of many commercially harvested bivalve species, such as *M. edulis*, *Mya arenaria*, and scallops (e.g., *Argopecten irradians*); their invasions have been correlated with declines in mollusc abundance and landings (Grosholz and Ruiz, 2002, Fofonoff et al., 2003, Klassen and Locke, 2007). Damage to shellfish populations in the Hudson Bay LME could harm harvested populations of bearded seal (*Erignathus barbatus*), Atlantic walrus (*Odobenus rosmarus*), and marine birds such as Hudson Bay eider (*Somateria mollissima*) that forage on benthic molluscs.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Carcinus maenas* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Carcinus maenas is native to Atlantic Europe, where it has been reported north to Iceland, and perhaps to northwest Africa (Carlton and Cohen, 2003, Compton et al., 2010). It has invaded temperate regions of Atlantic North America, Australia, South Africa, Japan, and Pacific North America. Thermal shifts (niche shifts) may be occurring that enable *C. maenas* to invade colder waters such as Nova Scotia and warmer waters such as Botany Bay, Australia (Compton et al., 2010, de Rivera et al., 2011). It was introduced to Svalbard in ballast water but has not been established (Ware et al., 2016).

Modelling suggests that invasion potential depends on the European origin, north or south, of the invasive population (Compton et al., 2010). Most invasions have arisen from southern

Europe and can be predicted based on annual summer and winter temperatures across the species' European range. This predictor does not work for invasions from northern Europe which are able to invade colder waters (e.g., Nova Scotia and Alaska). Green crabs on the Canadian east coast are a Norwegian phenotype; others have not invaded successfully (K.Howland, pers. comm.).

The species' range is expected to expand northward in response to climatic warming (Kelley et al., 2013). Ware et al. (2016) have suggested that habitat in southern James Bay may be suitable for the establishment of *C. maenas* and that under climatic warming scenarios all of James Bay and portions of southern Hudson Bay may become suitable by 2100. These projections may not take ice cover into account and may overestimate water column temperatures. Goldsmit et al. (2018) predicted that under global warming scenarios, and considering ice concentration, James Bay could be suitable for the green crab.

Chionoecetes opilio **(O. Fabricius, 1788)**

Phylum: Arthropoda
Class: Malacostraca
Order: Decapoda
Family: Oregoniidae

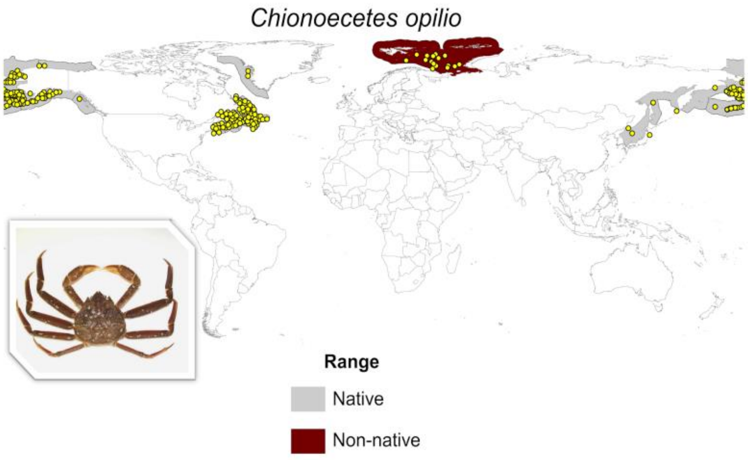

Figure 10: Ecoregions where Chionoecetes opilio is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Chionoecetes opilio occurrence points were obtained from OBIS (https://obis.org/), Hansen (2015), Burmeister and Sainte-Marie (2010). Picture of C. opilio modified from https://www.marinespecies.org/photogallery.php?album=717&pic=40630.

CMIST scores for *C. opilio:*

Mean adjusted Likelihood of Invasion: 2.58

Mean adjusted Impact of Invasion: 2.43

Mean adjusted Overall CMIST score: 6.28

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Chionoecetes opilio* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There are no reports of *C. opilio* found from Hudson Bay, (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016), though it is widely distributed on high/latitude continental shelves of the North Pacific and North Atlantic, and present in Alaskan waters and the Bering Sea (Hardy et al., 2011).

Q2- Rate of introduction **(Score = 3, Certainty = 2)**

Score Rationale: Some reliable information suggests *Chionoecetes opilio* could arrive frequently in high numbers in the RA area. The species has been found on arriving vessels and is present in a number of connected port ecoregions.

Background Information: No data were found on propagule pressure but these crabs are present in the Gulf of St Lawrence, West Greenland, and the Canadian Beaufort Sea, so the potential for transport of pelagic larvae in ballast water from these locations exists. In the Barents Sea it has been introduced by ballast (Alvsvåg et al., 2009). *Chionoecetes opilio* is widely distributed on high/latitude continental shelves of the North Atlantic (Squires, 1990) and North Pacific (Hardy et al., 2011). It occurs on Arctic coastal shelves well north of the HB LME:To the east it occurs on the Labrador Shelf north to the mouth of Hudson Strait (Squires, 1990) and along the west coast of Greenland north to at least Disko Bay (Burmeister and Sainte-Marie, 2010). To the west it has been reported from the Canadian Beaufort Sea east to Cape Parry (Squires, 1969, Divine et al., 2015). The presence of these crabs to the north could increase the likelihood of their natural dispersal into the HB LME.

Q3- Survival (suitable habitat) **(Score = 3, Certainty = 3)**

*Score Rationale: C*onsiderable reliable information suggests most of the RA area offers suitable habitat conditions for *Chionoecetes opilio*. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers (https://www.nceas.ucsb.edu/) to support habitat classification.

Background Information: Optimal depth ranges from 50-1400 m, but most *C. opilio* are found around 200 m, mainly in soft bottoms (Brockerhoff and McLay, 2011). Adult snow crabs are usually found from 0 to 450 m and temperatures from 0 to 5°C (Lovrich et al., 1995, Tremblay, 1997). Snow crab is usually associated with cold waters, but its quasi-absence in the Arctic ocean together with its bathymetric and geographic distribution characterize it as an arctic-boreal species (Williams, 1984, Squires, 1990). Benthic stages of snow crab take place on a soft bottom habitat exposed to water temperatures from –1.5 to 4 °C and salinity of > 26 ‰ (Sainte-Marie et al., 2005).

Q4- Survival (suitable environment) **(Score = 3, Certainty = 3)**

*Score Rationale: C*onsiderable reliable information suggests most of the RA area offers suitable environmental conditions for *Chionoecetes opilio*. Combination of temperature and salinity tolerance in most of the RA area. This is a species of cold water regions that could easily find the environmental conditions in the area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Benthic stages of snow crab take place on a soft bottom exposed to water temperature from –1.5 to 4°C and salinity of > 26 ‰ (Sainte-Marie et al., 2005). In the Barents Sea, the majority of crabs were found in waters colder than 2°C. It seems that higher water temperatures could be a limiting factor (Pavlov, 2006, Alvsvåg et al., 2009, Agnalt et al., 2011).

The new population established in the Barents Sea has been found at depths and temperatures similar to its natural habitat in the Northwest Atlantic and North Pacific. Smaller snow crabs seem to be more stenothermic, (i.e. less tolerant to slightly colder and slightly warmer temperatures than larger crabs), as is documented by Dionne et al. (2003) for snow crabs in the NW Gulf of St. Lawrence (Alvsvåg et al., 2009). The lethal temperature is 15°C but feeding stops and mortalities can occur in a short time at >12°C, especially in low salinity waters (Foyle et al., 1989, Hardy et al., 1994). *Chionoecetes opilio* tolerates salinities down to 21 ppt at 4°C and 28 ppt at 12°C (Hardy et al., 1994).

Q5- Establishment (reproductive requirements) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are almost always available for *Chionoecetes opilio* in the RA area. Most stages of reproduction can withstand cold environments within the range of the assessment area. The species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From Alvsvåg et al. (2009): “It is hypothesized that temperature is the main factor influencing incubation time. In areas where the temperature is from 2 to 4°C a 1-year cycle is more likely than in areas where the temperature ranges from -1 to 2°C (Comeau et al., 1999, Burmeister, 2001 and references therein).”

From Sainte-Marie et al. (2005): “During the larval phase, the best temperature for survival in the lab is from 7 to 15°C for zoea larvae and 10 to 15°C for megalopa larvae (Kon, 1980, Kogane et al., 2005). Immature crabs from the first benthic stages can tolerate cold temperatures from −1.8°C to 6°C during several months, but over the long term, their survival will decrease with temperatures ≤ –1°C or under 3°C (Thompson and Hawryluk, 1989, Gravel, 2002; B. Sainte-Marie and F. Hazel personal communication). Larvae can tolerate salinity variations, but only for short time periods. The benthic stage is stenohaline and normally associated with saline habitats over 26 ‰ (Williams, 1984, Squires, 1990).”

Q6- Establishment (natural control agents) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents could slow population growth in the RA area for *Chionoecetes opilio*. There is evidence that predation is likely an important control on populations in the assessment area. Although there are known parasitic biocontrol agents for crabs, there are none specific to this species and uncertainty regarding native parasites in the RA area. There is a wealth of information about predation and parasitism on snow crabs as well as crabs in general.

Background Information: From Brockerhoff and McLay (2011): “Parasites could act as a natural control agent in crabs, but as they have complex life cycles with intermediate hosts, the introduction of adult crabs together with their known parasites may be complicated. Native parasites in the new environment will not immediately attack the new host so its numbers can explode in the absence of any restraints (Brockerhoff and McLay, 2011). Ballast water may be one of the most successful vectors for aliens because larval stages cannot carry adult crab parasites (Torchin and Lafferty, 2009).”

From Hansen (2015): “Some of the relevant natural enemies that can be used in marine biocontrol of crabs are parasitic castrators (Lafferty, 1993) and symbiotic egg predators (Goddard et al., 2005), but there is not a known parasite that is specific only for the snow crab.”

Cod predation on juvenile benthic stages may exert control on snow crab abundance (Orensanz et al., 2004). In the Bering Sea this predation is mostly by small Pacific cod (*Gadus* *macrocephalus*) (Livingston, 1989) and in the Gulf of St Lawrence by Atlantic cod (*Gadus morhua*) (Bailey, 1982). The latter species is present in Hudson Strait (Stewart and Lockhart, 2005). Skates (especially thorny skate, *Raja radiata*—also in Hudson Strait), seals, flatfishes, squids, and other crabs are also important predators of these crabs in their native ranges (Choi (Orensanz et al., 2004, Choi and Zisserson, 2008). In Atlantic Canada, loss of cod and other benthic predators has been correlated with a substantial increase in snow crab abundance (Frank et al., 2005, Choi and Zisserson, 2008). These fish species are not present in the HB LME but related species might exert some control on the crab population. Density-dependent cannibalism may also exert some population control on *C. opilio* (Boudreau and Worm, 2012). In the Barents Sea, introduced *C. opilio* is preyed upon by Atlantic cod, haddock, wolfish (*Anarhichas* sp.) and thorny skates (*Raja* spp.) (Pavlov, 2006, Agnalt et al., 2011). Some of these species are present in the assessment area.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* has the capacity for a wide range of natural dispersal in the RA area.

Background Information: Chionoecetes opilio has a long larval life of 3–5 months (Puebla et al., 2008).

From Sainte-Marie et al. (2005): “Snow crab larvae can have a potential dispersion of hundreds of kilometers during the larval phase over 3 to 5 months (Lanteigne, 1986, Merkouris, 1988, Lovrich et al., 1995, Puebla, 2003). However, the various benthic stages generally have a much smaller dispersal capability (Lovrich et al., 1995).”

The species’ high fecundity (~5,000 to 160,000 eggs per female) (Davidson et al., 1985, Webb et al., 2016) give *C. opilio* the potential for natural dispersal over long distances. In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Chionoecetes opilio* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal based on shipping, through ballast in particular. Importance of other vectors such as biofouling is less clear and there is uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: Chionoecetes opilio has an established population in the Barents Sea, likely introduced by ballast water (Alvsvåg et al. 2009). The aquarium and restaurant trades utilize live imports (Ng 1998, from Brockerhoff and McLay 2011), though not prevalent in the Arctic. Ballast water transported by domestic coastal resupply and other boat traffic could spread *C. opilio* larvae from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland 2009; Chan et al. 2012). Chan (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* would have high impacts in many areas on population growth in the RA area. There is good evidence that this species eats a wide variety of prey and can impact their population growth.

Background Information: From Hansen (2015): “Snow crabs are generalist feeders known to feed on algae, mollusks, crustaceans, polychaetes, echinoderms and fish (Lovvorn, 2010). It is hypothesized that it can have the same level of impact as the red king crab *Paralithodes camtschaticus* (Hansen, 2015), which has been shown to reduce fauna (Oug et al., 2011) and can be a potential threat to the highly productive fisheries in the region through its ecosystem impacts (Falk-Petersen et al., 2011). Dvoretsky and Dvoretsky (2015) investigated whether red king crab and snow crab have had an effect on commercial fish and shrimp in the Barents Sea. They documented no negative impact on commercial fish populations, and no adverse effects to major fish stocks, while resulting in positive economic benefit. They did, however, not rule out negative effects on other parts of the ecosystem, and recognize that the snow crab is too new to see long-term effects.”

The pelagic larvae of snow crab are planktivores and the benthic juvenile and adult stages are generalist omnivores that prey upon or scavenge a wide variety of infauna and epifauna. In Atlantic Canada these include echinoderms (e.g., *Strongylocentrotus droebachinesis*, *Ophiura sarsi*, *Ophiopholis aculeata*), polychaetes (e.g., *Maldane* spp., *Nereis* spp.) and other worm-like invertebrates, plants, detritus, large zooplankton, shrimps, smaller crabs (*Hyas coarctatus*), bivalve molluscs (e.g., *Mytilus edulis*, *Hiatella arctica*), gastropods, sea anemones (*Metridium senile*), fishes (*Mallotus villosus*), and other taxa (Wieczorek and Hooper, 1995, Choi and Zisserson, 2008); Squires and Dawes 2003). Many of these species (e.g., those in brackets) occur in Hudson Bay (Stewart and Lockhart, 2005). They are also important predators of various commercially harvested echinoderm and mollusc species, and of capelin (*Mallotus villosus*) (Wieczorek and Hooper, 1995, Squires, 2003, Choi and Zisserson, 2008).

From Kaiser et al. (2018): "Potential elimination of some species caused by Snow Crab predation is particularly worrisome for Arctic ecosystems, due to potential severe effects both upwards and downwards in the food web, as opposed to other more resilient ecosystems where more diversity means some species' roles have more substitutability (Kaiser et al., 2018). Manushin (2016) finds that the snow crab has led to a decline in the biomass of benthic species, especially in the eastern parts of the Barents Sea where the crab abundance is high (Frantzen et al., 2017). In measurements of the benthos biomass for the Barents Sea, the decline observed after 2013 has been found to overlap with the maximum distribution of Snow Crab and the period of maximum benthos consumption (ICES, 2017). However, the area of reduced benthos also overlaps with an increase in bottom temperature (since 2007) and therefore the decline in benthos biomass is attributed to multiple impact factors (ICES, 2017), which makes it hard to disentangle the effects of the Snow Crab alone.

Direct and indirect threats to other commercial fisheries are also possible. In Newfoundland, the Snow Crab has been observed to feed on capelin (Squires, 2003) and thus there might be a possibility of the crab representing a threat for post-spawn capelin (Mikkelsen, 2013). Other concerns include the food competition the crab might create with other commercially valuable species, through its predation on the Polychaeta community. The northern shrimp is an example of a commercially exploited species whose distribution is partly overlapping with the Snow Crab distribution, mainly in Norwegian waters of the Barents. Dvoretsky and Dvoretsky (2015) find a negative correlation between biomass of Snow Crabs and shrimp, which they attribute to prey-predator interactions.”

Cannibalism is common in *C. opilio* (Wieczorek and Hooper, 1995) and cods (e.g., *Gadus morhua*), skates (e.g., *Raja radiata*), seals, flatfishes, squids, and other crabs are important predators in its native range (Orensanz et al., 2004, Choi and Zisserson, 2008). In its introduced Barents Sea range, Atlantic cod, haddock, wolffish (*Anarhichas* spp.) and thorny skates (*Raja* spp.) prey on them (Pavlov, 2006, Agnalt et al., 2011). It is possible that feeding on *C. opilio* may increase productivity in these species.

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* would have high impacts in many areas on communities in the RA area. There is good evidence that this species eats a wide variety of prey and can heavily impact benthic communities.

Background Information: Chionoecetes opilio creates competition for food with other crabs and modification of the food web (Wieczorek and Hooper, 1995). They consume mainly polychaetes and bivalves, however, the most important prey biomass is shrimp and fish (Boudreau and Worm, 2012).

Quijón and Snelgrove (2005a,b) have shown that predation by snow crabs has a marked effect on infaunal benthic communities.

From Hansen (2015): "It is hypothesized that snow crab can have the same level of impact as the red king crab *P. camtschaticus* (Hansen, 2015), which has been shown to reduce fauna and affect structural and functional diversity (Oug et al., 2011). Snow crabs are generalist feeders known to feed on algae, mollusks, crustaceans, polychaetes, echinoderms and fish (Lovvorn, 2010)."

Adult stages are generalist omnivores that prey upon or scavenge a wide variety of infauna and epifauna. In Atlantic Canada these include echinoderms (e.g., *Strongylocentrotus droebachinesis*, *Ophiura sarsi*, *Ophiopholis aculeata*), polychaetes (e.g., *Maldane* spp., *Nereis* spp.) and other worm-like invertebrates, plants, detritus, large zooplankton, shrimps, smaller crabs (*Hyas coarctatus*), bivalve molluscs (e.g., *Mytilus edulis*, *Hiatella arctica*), gastropods, sea anemones (*Metridium senile*), fishes (*Mallotus villosus*), and other taxa (Wieczorek and Hooper, 1995, Choi and Zisserson, 2008); Squires and Dawes 2003). Many of these species (e.g., those in brackets) occur in Hudson Bay (Stewart and Lockhart, 2005).

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* would have high impact in many areas on habitat in the RA area. There is extensive information about large decopods digging in and disturbing benthic sediments.

Background Information: It is hypothesized that snow crab can have the same level of impact as the red king crab *P. camtschaticus* (Hansen, 2015), which has been shown to make changes in sediment integrity (Oug et al., 2011) and structure due to their foraging activities (Boudreau and Worm, 2012). Crabs also burrow into sediments (Miller, 1975, Conan and Maynard, 1987) and Bernard Sainte-Marie (personal communication) has described watching them dig to hunt polychaetes and the like, tearing up the bottom in the process. Wieczorek and Hooper (1995) observed snow crabs sticking their chelae into sediments like forceps to feed, but that they do not seem to "dig".

The crabs themselves are also physical structures and may represent new habitats that could allow increased biodiversity (Falk-Petersen et al., 2011). Representatives of at least 21 sessile invertebrate families have been found on the exoskeleton of *C. opilio* from the Scotian Shelf to southern Gulf of St. Lawrence (Savoie et al., 2007).

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* would have high impacts in many areas on ecosystem function in the RA area. There is good evidence that this species has substantial ecosystem-level impacts.

Background Information: From Hansen (2015): "It is hypothesized that snow crab can have the same level of impact as the red king crab *P. camtschaticus* (Hansen, 2015), where it has been shown to affect structural and functional diversity (Oug et al., 2011). Its omnivorous characteristics (see rationale Q. 9 and 10) could disrupt food-web structure - invasive crustaceans are often the causes of substantial impact due to their omnivorous role leading to shifts in energy fluxes, nutrient cycles and thus, affect critical ecosystem services, biodiversity and fisheries (Hänfling et al., 2011)."

From Kaiser et al. (2018): Potential elimination of some species caused by Snow Crab predation is particularly worrisome for Arctic ecosystems, due to potential severe effects both upwards and downwards in the food web, as opposed to other more resilient ecosystems where more diversity means some species' roles have more substitutability. Echinoderms for example, which are known to play an important role in redistribution and remineralization of organic carbon on Arctic shelves (Bluhm et al., 2009), are among the most common prey items for the Snow Crab (Jørgensen and Spiridonov, 2013). Foraminifera, another dominant category of prey species found in stomach contents of the Snow Crab (Jørgensen and Spiridonov, 2013), represent an important link between lower and higher levels of the food web (Gooday et al., 1992, Hansen, 2015). Changes in the abundance and composition of echinoderms may lead to large structural ecosystem changes and so do potential changes in foraminifera (Hansen, 2015).

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Chionoecetes opilio* would have low or no impact associated with diseases, parasites or travelers in the RA area. The main vector of introduction for *C. opilio* is ballast water containing the larvae stage, hence all parasites and diseases that can be transported by adult crabs would be very low. However, there may be a possibility of expansion into the RA area by movement of adults (with associated parasites and diseases) established in adjacent ecoregions (Labrador) which would add some uncertainty as to whether the risk could be higher.

Background Information: From Hansen (2015): "Hydrozoans, bryozoans, polychaetes, and acorn barnacles are commonly found on large crustaceans as epibionts (Savoie et al., 2007). Parasitic dinoflagellates of *Hematodinium* spp are the main pathogens found on snow crab, causing bitter crab disease (Hansen, 2015). This disease can kill the crab but also causes the crabmeat to have a bitter flavor. Prevalence and distribution of the parasite and its controlling factors are poorly understood, although it seems to be density regulated (Hansen, 2015)."

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: There are no native crabs in the same genus, so *C. opilio* is unable to hybridize with local species (https://obis.org/). Natural hybridization can occur between *Chionoecetes japonicus* and *C. opilio*, but only with *C. japonicus* as the maternal parent (Kim et al., 2012).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Chionoecetes opilio* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. Some benthic species at risk could be impacted given the demonstrated effects of this crab species on benthic ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and the at-risk bird species are unlikely to be affected. There are several at risk or special concern species of fish that could be affected through competition of predation - these include: northern wolffish, *Anarhichas denticulatus*; spotted Wolffish, *Anarhichas minor*; thorny Skate, *Amblyraja radiata*. Walrus populations of risk/special concern may also be impacted as they feed on benthic invertebrates so may be impacted through competition for food resources or disturbance of benthic habitats by the snow crab. Beluga whales (*Delphinapterus leucas*) (COSEWIC 2004), and perhaps other marine mammal species might benefit from *C. opilio* by preying on them, but could also see a reduction in their other prey through direct or indirect competition.

Q16- Impact on fisheries **(Score = 3, Certainty = 2)**

Score Rationale: Some reliable information suggests *Chionoecetes opilio* would have high impacts in many areas on fished species in the RA area. There are well demonstrated impacts on abundances of harvested species elsewhere and high potential for impacts in the RA area. There have been mixed findings that may be confounded by concurrent shifts in environmental conditions that also could have effects on species abundance/composition.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries along the Nunavik coast; blue mussel (*Mytilus edulis*), green sea urchin (*Strongylocentrotus droebachiensis*), brown sea cucumber (*Cucumaria japonica*) and six-rayed starfish (*Leptasterias polaris*) have been harvested by commercial test fisheries in the Belcher Islands (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Commercial fisheries for shrimp are found in Hudson Strait and may be impacted by Snow Crab through food web/ecosystem changes (DFO 2018). Snow crabs are important predators of various commercially harvested echinoderm and mollusc species, and of capelin (*Mallotus villosus*) (Wieczorek and Hooper, 1995, Squires, 2003, Choi and Zisserson, 2008). These crabs could become bycatch (Brockerhoff and McLay, 2011) of coastal commercial and subsistence gillnet fisheries for anadromous fishes, particularly Arctic charr (*Salvelinus alpinus*) (Stewart and Lockhart, 2005), and cause damage to the fish and nets. Damage to shellfish populations in the Hudson Bay LME could harm harvested populations of bearded seal (*Erignathus barbatus*), Atlantic walrus (*Odobenus rosmarus*), and marine birds such as Hudson Bay eider (*Somateria mollissima*) that forage on benthic molluscs and other invertebrates. Some seals and whales (e.g., belugas and narwhals) may offset these impacts by eating these large crabs instead.

From Hansen (2015): “It is hypothesized that snow crab can have the same level of impact as the red king crab *P. camtschaticus* (Hansen, 2015), which can be a potential threat to the highly productive fisheries in the Barents Sea through its ecosystem impacts (Falk-Petersen et al., 2011). Dvoretsky and Dvoretsky (2015) investigated whether red king crab and snow crab have had an effect on commercial fish and shrimp in the Barents Sea. They documented no negative impact on commercial fish populations, and no adverse effects to major fish stocks, while resulting in positive economic benefit. They did, however, not rule out negative effects on other parts of the ecosystem, and recognize that the snow crab is too new to see long-term effects.”

From Kaiser et al. (2018): "Potential elimination of some species caused by Snow Crab predation is particularly worrisome for Arctic ecosystems, due to potential severe effects both upwards and downwards in the food web, as opposed to other more resilient ecosystems where more diversity means some species' roles have more substitutability (Kaiser et al., 2018). Manushin (2016) finds that the Snow Crab has led to a decline in the biomass of benthic species, especially in the eastern parts of the Barents Sea where the crab abundance is high (Frantzen et al., 2017). In measurements of the benthos biomass for the Barents Sea, the decline observed after 2013 has been found to overlap with the maximum distribution of Snow Crab and the period of maximum benthos consumption (ICES, 2017). However, the area of reduced benthos also overlaps with an increase in bottom temperature (since 2007) and therefore the decline in benthos biomass is attributed to multiple impact factors (ICES, 2017), which makes it hard to disentangle the effects of the Snow Crab alone.

Direct and indirect threats to other commercial fisheries are also possible. In Newfoundland, the Snow Crab has been observed to feed on capelin (Squires, 2003) and thus there might be a possibility of the crab representing a threat for post-spawn capelin (Mikkelsen, 2013). Other concerns include the food competition the crab might create with other commercially valuable species, through its predation on the Polychaeta community. The northern shrimp is an example of a commercially exploited species whose distribution is partly overlapping with the Snow Crab distribution, mainly in Norwegian waters of the Barents. Dvoretsky and Dvoretsky (2015) find a negative correlation between biomass of Snow Crabs and shrimp, which they attribute to prey-predator interactions."

Q17- Past invasion history **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Chionoecetes opilio* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Chionoecetes opilio is a Sub-Arctic species found in the North Pacific, Beaufort Sea, northwest Atlantic oceans and west coast of Greenland. The invaded range, first recorded in 1996, is within the Barents Sea and was presumably introduced by ballast water (Alvsvåg et al., 2009, Brockerhoff and McLay, 2011). In the Barents Sea the population was found at depths, temperature range and substrates similar to its native habitat (Agnalt et al., 2011).

Eriocheir sinensis H. Milne Edwards, 1853

Phylum: Arthropoda
Class: Malacostraca
Order: Decapoda
Family: Varunidae

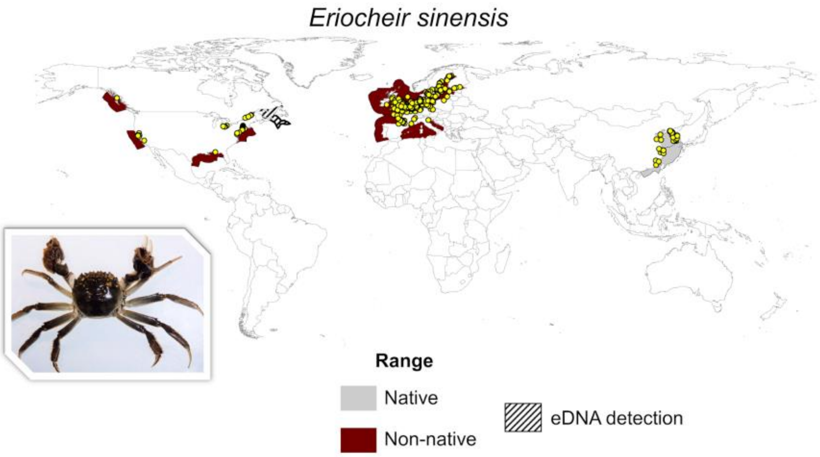

Figure 11: Ecoregions where Eriocheir sinensis is distributed: native (grey), non-native (dark red) regions, and only detected through e-DNA samples (black and white stripes). These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected

in an exhaustive manner. Eriocheir sinensis occurrence points were obtained from OBIS (https://obis.org/). Note that this species can also be present in freshwater ecosystems, therefore points on land were left on the map. (As explained in Question 17, E. sinensis has been introduced along the Atlantic coast and in the Great Lakes of North America but may not be established). Picture of E. sinensis modified from https://invasions.si.edu/nemesis/species_summary/99058.

CMIST scores for *E. sinensis:*

Mean adjusted Likelihood of Invasion: 1.95

Mean adjusted Impact of Invasion: 2.31

Mean adjusted Overall CMIST score: 4.50

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Eriocheir sinensis* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There were no reports of *E. sinensis* found in Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). It was first detected in North America in 1965 in the Detroit River at Windsor in the Great Lakes (Veilleux and De Lafontaine, 2007). Three specimens taken from the St. Lawrence estuary in 2006 provided the first evidence of these crabs in suitable reproductive brackish waters within the Great Lakes–St. Lawrence River basin. There are now confirmed reports of breeding females in Chesapeake and Delaware bays on the east coast of the USA (Dittel and Epifanio, 2009). Specimens have been collected from the Columbia River estuary, Oregon, but are not necessarily established. They were reported in San Francisco Bay, California in 1992 and have been well established there since the mid-1990s (Rudnick et al., 2000).

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eriocheir sinensis* could arrive frequently in low numbers or infrequently in high numbers in the RA area. They are widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Ballast water is the most likely transport vector for introduction of *E. sinensis* to Hudson Bay. Propagule pressure is unknown but *E. sinensis* is present at ports directly connected to Churchill and Deception Bay by ships travelling in ballast (Chan et al., 2012). Therriault et al. (2008a) considered ballast water the single most important vector for the initial establishment of Chinese mitten crabs, followed by intentional release and aquaculture, with adult migration and larval drift as the most important vectors for secondary dispersal. Holeck et al. (2004) reported the presence of active *E. sinensis* associated with sediment in vessels with (BOB) and without (NBOB) ballast on board that entered the Great Lakes.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Eriocheir sinensis*. The species requires muddy banks in estuaries which only occur in a moderate proportion of the RA. Although there is good information on the species' requirements, we do not have complete knowledge of all substrate types in the estuaries within the RA area.

Background Information: Eriocheir sinensis is typically found in estuaries and the lower parts of rivers (Veilleux and De Lafontaine, 2007). Estuaries supporting large mitten crab populations are characterized by large areas of brackish waters for embryonic and larval development, as well as large areas of shallow productive waters for the growth of juveniles (Cohen and

Weinstein, 2001). The Yangtze River, for example, which is one of the major rivers used by mitten crab in its native China, is characterized by long freshwater drainage with warm, slow moving water and a large estuary.

The tendency of this species to undergo rapid, population-wide fluctuations in abundance suggests that environmental parameters play a strong role in governing the crab's population dynamics (Rudnick et al., 2005a).

Hudson and James bays receive inflow from many rivers and have numerous large estuaries (Stewart and Lockhart, 2005). The Port of Churchill is located adjacent to the Churchill River estuary. The Nelson River estuary to the east (i.e., downstream) is particularly large and shallow; current in the river fluctuates through the day in response to hydroelectric demand. *Eriocheir sinensis* can move upstream against currents of up to 1.5 m/s (i.e., 5.6 km/h) (Cohen and Weinstein, 2001).

Adult Chinese mitten crabs utilize fresh, brackish, and saltwater habitats, constructing burrows in riverbanks and levees, intertidal areas, and marsh bottoms. (https://wildlife.ca.gov/Conservation/Invasives/Species/Mitten-Crab; http://nyis.info/invasive_species/chinese-mitten-crab/)

Q4- Survival (suitable environment) (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a negligible proportion of suitable environmental conditions for *Eriocheir sinensis.* Based on modelling and published temperature tolerances there is not much of the RA area that matches the species requirements. Species requirements are well known and there are reliable environmental data layers and modelling to show a lack of suitable habitat in the RA area.

Background Information: Mitten crabs can mature in freshwater, but require brackish or salt water for reproduction and development (Cohen and Weinstein, 2001). Changes in their salinity and temperature requirements during their life-cycle are likely an adaptation to patterns of larval dispersal: (1) hatching in brackish water of outer estuaries; (2) offshore (near-surface) transport during zoeal development (5 stages), with late stages living mainly under marine conditions; (3) onshore (near-bottom) transport of the megalopa; (4) settlement in any part of an estuary, probably most frequently at ca. 15 to 25% ppt (Anger, 1991). While metamorphosis and settlement from the megalops stage, juvenile and adult growth, and gonadal development can occur in freshwater, some minimum levels of salinity are required for reproduction and embryonic and larval development (Cohen and Weinstein, 2001). In California, young juveniles (<10 mm) are found in tidally influenced, low salinity (1–10%) habitats (Rudnick et al., 2003, Rudnick et al., 2005a) and migrate into freshwater to mature into adults. Modelling results suggest a northern limit of suitable habitat in southern Canada (Herborg et al., 2007, Therriault et al., 2008a).

Mitten crabs are present in Lake Ladoga, Russia, where winter conditions are icy and water temperatures average <2°C (range >0° to 4°C) from December through April (Panov, 2006, Naumenko et al., 2007). These crabs can be abundant in river systems with typical winter temperatures in the estuary as low as 5-10°C, and summer temperatures of up to 23-25°C (Cohen and Weinstein, 2001). Adult mitten crabs can maintain a positive energy balance at salinities from 0.5 to 25 ppt (= range tested, held at 15°C) (Normant et al., 2012) and temperatures of 7° and 17°C (= range tested, held at 0.5 ppt S) (Jakubowska and Normant, 2011).

Q5- Establishment (reproductive requirements) (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are almost never available for *Eriocheir sinensis* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Minimum sea surface salinity <=25 ppt and maximum sea surface temperature >=12 °C (published threshold values for reproduction and larval development) resulted in a negligible amount of the RA area that would provide suitable habitat for the completion of their life cycle. The species requirements are well demonstrated and there are reliable environmental data layers and modelling to show a lack of suitable habitat in the RA area.

Background Information: Mitten crabs require brackish or salt water for reproduction and development (Cohen and Weinstein, 2001). Changes in their salinity and temperature requirements during their life-cycle are likely an adaptation to patterns of larval dispersal: (1) hatching in brackish water of outer estuaries; (2) offshore (near-surface) transport during zoeal development (5 stages), with late stages living mainly under marine conditions; (3) onshore (near-bottom) transport of the megalopa; (4) settlement in any part of an estuary, probably most frequently at ca. 15 to 25% ppt S (Anger, 1991). While metamorphosis and settlement from the megalops stage, juvenile and adult growth, and gonadal development can occur in freshwater, some minimum levels of salinity are required for reproduction and embryonic and larval development (Cohen and Weinstein, 2001). In California, young juveniles (<10 mm) are found in tidally influenced, low salinity (1–10%) habitats (Rudnick et al., 2003, Rudnick et al., 2005a) and migrate into freshwater to mature into adults.

Mating appears to require salinities of > ca. 5-10 ppt (optimum ~15 ppt); egg adherence > ca.15 ppt, possibly higher (optimum ca. 25-30 ppt); embryonic development and hatching > ca. 5-10 ppt (optimum ca. 15-25 ppt); zoeal development > ca. 10-15 ppt (optimum >15 or 20 ppt) (Cohen and Weinstein, 2001). Intermediate zoeal stages require at least 16–17 ppt salinity to survive in the laboratory (Anger, 1991). For aquaculture, the optimal salinity for egg incubation and larval rearing is 20–25 ppt (Sui et al., 2011). Some of the stages between mating and megalopa may have upper salinity limits of ca. 25-35 ppt (Cohen and Weinstein, 2001). Large estuaries with well-developed estuarine circulation may provide optimal salinities for larval development. Development time increases at unfavorably low or high salinities (Anger, 1991).

Estuary temperatures of ca. 15-25°C in the spring and summer are needed for good hatching success and larval development. Optimal hatching for aquaculture is at 18-21°C (Sui et al., 2011). In the laboratory, successful development from hatching to metamorphosis occurred at ≥12°C, with no survival of the first zoea stage at 9°C (Anger, 1991). Temperatures of 15-30°C are needed in the upper estuary and river for good juvenile and adult growth but they can survive at 4° to 31-32°C (Cohen and Weinstein, 2001). Optimal conditions for mating and spawning of aquaculture brood crabs is 13–17 ppt and 9–13°C (Sui et al., 2011). Overall survival and salinity tolerance increased with increasing temperature (range tested 6°-18°C, but no survival of first zoea stage below 9°C), whereas development duration decreased exponentially (Anger, 1991). At 12°C and 25 ppt S development took 90 days (Anger, 1991, Rudnick et al., 2005a).

Q6- Establishment (natural control agents) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests natural control agents are unlikely to affect population growth in the RA area for *Eriocheir sinensis.* Although there is considerable evidence that predation can occur on this species (including species found in the RA area), the available information indicates it is unlikely to curtail population growth. Although there are known parasitic biocontrol agents for crabs, there are none in the RA area. Despite knowing much about the species' biology, it is unclear the extent to which population growth is curtailed by predators and parasites.

Background Information: Predation is not considered a primary factor controlling population size, except in China (Hanson and Sytsma, 2005), where ovigerous female crabs are harvested for their eggs (Hymanson et al., 1999). The mitten crab is protected from predation by its large size, protective shell, nocturnal habits, and ability to create and use burrows as refugia. In areas such as Germany and San Francisco Bay these invasive crabs are preyed upon by a variety of fish, wading birds, and mammals (Panning, 1939, Rudnick et al., 2000). Their continued abundance indicates that natural control has not successfully limited these crab populations.

From Veilleux and De Lafontaine (2007): "Little is known about the predators of the Chinese mitten crab. It was suggested that many fish species including pike (*Esox* spp.), eels (*Anguilla* spp.), brown trout (*Salmo trutta*), sturgeon (*Acipenser* spp.), striped bass (*Morone saxatilis*) and channel catfish (*Ictalurus punctatus*) could prey upon the crab. Other aquatic related animals such as bullfrogs, raccoons, river otters, wading birds and humans may also be counted among the potential predators of the mitten crab (Hymanson et al., 1999, Veldhuizen and Stanish, 1999, Veldhuizen and Hieb, 1998, cited in Hanson and Sytsma, 2005)."

From Bouma and Soes (2010): “Humans are most likely to be the most important predator of Chinese mitten crabs. In China for example, they are a traditional food source and an important part of the aquaculture industry, yielding a high annual production (Hymanson et al., 1999). In The Netherlands they are consumed on a small scale by the Chinese minority. Apart from humans the Chinese mitten crab has many other natural predators. The following overview of known predators in Germany was given by Fladung (2000): mammals, brown rat, black rat, polecat and river otter; birds: grey heron, stork, ducks (including Red-breasted Merganser), crows and seagulls; fish: eels, brown trout, common barb, perch, golden orfe, cod, pike, ruffe, elongate freshwater cod, pike-perch and big bream. Juvenile and small adult crabs are likely mostly targeted by these predators. In The Netherlands it is likely that these smaller individuals are eaten by bigger fish species feeding on benthic organisms such as eels, perch and roach (Soes et al., 2007). Crabs with a carapace width of 7 cm or more can probably only be consumed by large pike and Welsh catfish, both not (yet) very abundant in The Netherlands (Soes et al., 2007)”

Potential predators of mitten crabs in the rivers and estuaries around Hudson Bay include sturgeons (*Acipenser fulvescens*), various shorebirds, river otters (*Lontra canadensis*) and humans.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* has the capacity for a wide range of natural dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal based on their extended planktonic larval stage, ability to migrate long distances as adults and history of spread elsewhere.

Background Information: Eriocheir sinensis is capable of rapid, extensive invasions of coastal areas and freshwater tributaries (Panning, 1939, Rudnick et al., 2000, Herborg et al., 2003, Veilleux and De Lafontaine, 2007). Larval drift is likely an important mechanism for coastal spread (Therriault et al., 2008a). In the laboratory the larvae can remain planktonic for 93 days at 12°C and 25 ppt S (Anger, 1991). Mechanisms for their retention in their natal estuary are uncertain (Dittel and Epifanio, 2009). Over time, alongshore currents (Granskog et al., 2011) might disperse larvae counter-clockwise around the Hudson/James bay coasts, from estuary to estuary, and then east along the south coast of Hudson Strait. After settling and growing to a carapace width of about 20 mm (Dittel and Epifanio, 2009) the juveniles migrate into freshwater, where they can penetrate many hundreds of km upstream into freshwater tributaries, walking on land to bypass barriers (Panning, 1939, Herborg et al., 2003).

Information is needed on the spatial distribution of larvae in estuaries and on their vertical migrations to better understand the potential for larval dispersal from estuaries (Dittel and Epifanio, 2009).

It is now largely accepted that the worldwide spread of the Chinese mitten crab was due to human-mediated activities and not the result of natural causes (Cohen and Carlton, 1997). These authors identified 10 pathways (intentional and unintentional) that would explain the introduction and transfer of the crab around the world: i) dispersal of larvae by currents, ii) passive dispersal of adults or juveniles on floating material, iii) transport of adults or juveniles by ship fouling, iv) transport of adults or juveniles in cargo, v) transport of adults or juveniles on semi-submersible drilling platforms, barges and other long-distance slow-moving vessels, vi) transport of larvae or juveniles in ballast water, vii) transport of adults or juveniles in fisheries products, viii) transport of larvae in water with shipments of live fish, ix) escape or release from research, public, or private aquaria, x) intentional transfer to develop a food resource. Among all these vectors, two were considered to be the most likely pathways: the active transport and voluntary release of mitten crabs into new habitats to provide a new human food source, and the accidental release of crabs via ship ballast water discharge (Cohen and Carlton, 1997).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eriocheir sinensis* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal based on shipping, through ballast in particular. Importance of other vectors such as biofouling is less clear and some uncertainty as to the amount of ballast currently being uptaken, transported and released within the RA area.

Background Information: Ballast water transported by domestic coastal resupply and other boat traffic, and possibly hull fouling, could spread *E. sinensis* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of total) in the region, suggesting that this is a plausible vector for within-region spread. With the exception of ship-related movements, the importance of other types of movements (e.g., fishing vessels) in HBC is not well known.

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Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* would have high impacts in many areas on population growth in the RA area. This is a well-known invader that causes great impacts to a variety of populations and communities.

Background Information: Eriocheir sinensis are opportunistic omnivores that can become abundant and can occur in very high densities during migration (Panning, 1939, Normant et al., 2002). As invaders, especially to freshwater rivers that lack crabs, they are likely to exert new pressures on autochthonous food sources and shallow-dwelling invertebrates and could alter the structure of benthic communities, possibly causing or contributing to trophic cascades (Normant et al., 2002, Rudnick and Resh, 2005, Rosewarne et al., 2016). The crab's wide-ranging foraging techniques, use of intertidal habitat, and migration out of freshwater at sexual maturity increases the distribution of these impacts. They affect other species through competition for prey and habitat. Competition with native crayfish in rivers and other crustaceans in estuaries is likely.

From NEMESIS: "*Eriocheir sinensis* is an opportunistic omnivore. Juvenile and adult crabs feed on detritus, algae, aquatic plants, invertebrates, and dead or trapped fishes. Stable isotope, gut contents, and feeding studies in San Francisco Bay indicate that this species feeds heavily on aquatic derived detritus, algae, and invertebrates feeding on the sediment surface. Dense populations of mitten crabs could alter stream communities by decreasing the abundance of surface-dwelling invertebrates, and shifting invertebrate populations to deeper dwelling infauna (Hymanson et al., 1999, Rudnick et al., 2000, Rudnick and Resh, 2005). Mitten crabs are not normally fish predators, since they are slow-moving, but in confined spaces such as fishways, and holding-tanks in power plants and irrigation systems, they can attack fish or cause mortality through sheer crowding and clogging of passageways. In California, they caused serious mortality in fish-salvage facilities, designed to divert fish from irrigation facilities in the Sacramento-San Joaquin Delta. At peak migration times, mortality of migrating fish in 1998 was 98-99% (Rudnick et al., 2000) (Chinese Mitten Crab Control Committee 2003). Direct predation on fish eggs in spawning streams (for example, to Steelhead Trout (*Oncorhynchus mykiss*)), is a concern in the San Francisco Bay area (Chinese Mitten Crab Control Committee 2003).

Chinese Mitten Crabs may potentially compete with other crustaceans in estuarine and freshwater habitats for food and space. The diet of the Blue Crab (*Callinectes sapidus*) is more carnivorous (Lippson and Lippson, 1997), but both species are opportunistic, so a high density of Mitten Crabs could result in competition for invertebrate food and carrion. In freshwater, competition for food with native and introduced crayfish is possible. Crayfish are also omnivorous, so their food supply could be affected by the introduction of mitten crabs. Mitten Crabs have a feeding rate up to 3 times higher than that of the European Crayfish *Austropotamobius pallipes* in experiments, and so could compete with native crayfish and have a larger predatory impact on native benthic communities (Taylor and Dunn, 2018).

Shelter from predators can be critical for crabs and crayfish, especially juveniles. Juvenile Mitten Crabs are potential competitors for hiding places under rocks and logs and in submerged vegetation (SAV) beds (Rudnick et al., 2000). In brackish estuarine waters on the East Coast, such competition is likely with mud crabs (e.g. *Rhithropanopeus harrisii* (Harris Mud Crab), *Eurypanopeus depressus* (Flatback Mud Crab)), and with juvenile Blue Crabs. In fresh waters, introduced crayfish frequently displace native species from shelters through aggressive behavior (Bovbjerg, 1970). Juvenile Mitten Crabs dig burrows in riverbanks, and could compete with other burrowing species, such as *Uca minax* (Red-Jointed Fiddler Crab) in brackish areas, and *Cambarus diogenes* (Devil Crayfish) in fresh waters (Lippson and Lippson, 1997). In the Thames estuary, England, and in laboratory experiments, juvenile *E. sinensis* excluded native *Carcinus maenas* (Green Crabs) of similar size from shelters under boulders (Gilbey et al., 2008)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=99058)

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* would have high impacts in many areas on communities in the RA area. This is a well-known invader that causes great impacts to a variety of populations and communities.

Background Information: Predation of salmonid eggs and larvae (e.g., Arctic char *Salvelinus alpinus* and Brook trout *S. fontinalis*) is possible (Culver, 2005).

Eriocheir sinensis are opportunistic omnivores that can become abundant and can occur in very high densities during migration (Panning, 1939, Normant et al., 2002). As invaders, especially to freshwater rivers that lack crabs, they are likely to exert new pressures on autochthonous food sources and shallow-dwelling invertebrates and could alter the structure of benthic communities, possibly causing or contributing to trophic cascades (Normant et al., 2002, Rudnick and Resh, 2005, Rosewarne et al., 2016). The crab's wide-ranging foraging techniques, use of intertidal habitat, and migration out of freshwater at sexual maturity increases the distribution of these impacts. They affect other species through competition for prey and habitat. Competition with native crayfish in rivers and other crustaceans in estuaries is likely.

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Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* would have high impacts in many areas on habitat in the RA area. Impacts may be great within the species' suitable habitat.

Background Information: In areas under tidal influence *E. sinensis* digs burrows into the banks to create water-filled refuges during low tide (Panning, 1939, Rudnick et al., 2000, Herborg et al., 2003). There can be up to 30 burrows per sq. meter. Many of the rivers around Hudson and James bays are influenced by hydroelectric development and, depending upon the operating regime, have fluctuating water levels. In the Nelson River, for example, the water level can fluctuate up to 3 m daily in response to changing demand for electricity (D.B. Stewart, pers. obs.). The river mainstream and its tributaries have steep sand and clay banks that are already subject to erosion by the fluctuating water levels and ice scour, both related to hydroelectric development. If these "hydro tides" were to prompt extensive burrowing by the crabs, bank stability would be further reduced and erosion and water turbidity increased with broad effects on aquatic ecosystems in the river and estuary.

From NEMESIS: “In their native habitat and in California and Europe, juvenile *Eriocheir sinensis* create extensive burrows in the banks of the tidal portions of streams (Rudnick et al., 2005b). This extensive burrowing in steep creek banks creates increased erosion, slumping, and sometimes collapse of river banks. In man-made levee and dike systems, mitten crab burrows can increase the frequency of catastrophic flooding during rain events and contribute to the loss of tidal wetlands. Burrows in South San Francisco Bay tidal streams ranged from simple tube-like holes to complex, many-branched systems with multiple entrances, but all sloped downward from the entrance, so that they retained water after high tide. In different areas, total burrow volume averaged 700-900 cm^3 (Rudnick et al., 2005b). Burrowing crabs ranged from 4 mm to 45 mm carapace width, and burrow diameter was about twice the crab's carapace width. Overall, in the study area, burrows removed from 0.8 to 5.7% of the stream bank sediment during the study (Rudnick et al., 2005b). In mesocosm experiments, Mitten Crabs can remove plant shoots of Eurasian Watermilfoil (*Myriophyllum spicatum*), mostly by movement, and by aggressive interactions between crabs, with only minor levels of direct herbivory. High densities of mitten crabs can lead to complete vegetation loss (Schoelynck et al., 2020).” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=99058)

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* would have high impacts in many areas on ecosystem function in the RA area. The species is a well-known ecosystem engineer, altering nearshore environments.

Background Information: Burrowing can increase turbidity and nutrient exchange, and increase riverbank erosion (Panning, 1939, Rudnick et al., 2000, Herborg et al., 2003). Migration out of freshwater systems to reproduce and die in estuaries may constitute a substantial vehicle for exporting biomass, which may impact the food web, particularly with large densities of crabs (Rudnick and Resh, 2005). Crab abundance may be cyclical and these migrations can be much larger in some years than others (Panning, 1939).

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Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* would have low or no impact associated with diseases, parasites or travelers in the RA area. The main vector of introduction of *E. sinensis* is ballast water containing the larvae stage, hence all parasites and diseases that can be transported by adult crabs would be very low. There is little possibility of expansion into the RA area by movement of adults (with associated parasites and diseases) established in other regions (Quebec).

Background Information: Numerous invertebrate epibionts (mainly nematodes, bivalves and crustaceans) inhabit the dense setal mats covering the claws of *E. sinensis* (Normant et al., 2007).

Eriocheir sinensis has been identified as one of the second intermediate hosts (others including *Potomon* spp, *Sinopotamon* spp., etc.) for the oriental lung fluke, *Paragonimus westermani.* The fluke requires one of a variety of snail species (e.g., *Brotia* spp., *Melanoides* sp., *Semisulcospira* spp.) as first intermediate host and mammals comprise the final host, including humans who consume under-cooked crab (WHO, 1995; http://www.cdc.gov/parasites/paragonimus/biology.html). However, *E. sinensis* may only be an occasional host of the fluke (Hymanson et al., 1999, Dugan, 2002, Cohen, 2003). The fluke has been reported from a crab in Canada, presumably in shipments of live crabs from China (Hymanson et al., 1999), but not in extensive testing of mitten crabs in the San Francisco Bay area (Dugan, 2002). Several snail species that can act as second intermediate hosts of the parasite have apparently been introduced in the latter area, so if the fluke is introduced there and able to use *E. sinensis* as a primary host it might establish. If established, this fluke would pose a significant human health risk (Cohen, 2003). While the intermediate snail species have not been reported in the HB LME, it is not known whether native snails could act as the flukes' first intermediate host.

From Bouma and Soes (2010): "Several diseases have been described affecting the Chinese mitten crab including the parasitic barnacle *Polyascus gregaria*, epiphytic parasites, viruses and bacteria. The parasitic barnacle is only reported within the native range of the Chinese mitten crab. This barnacle affects the reproductive organs of the crabs. All epiphytic parasites known from the Chinese mitten crab belong to the genera *Zoothamnium*, *Vorticella* and *Epistylis*. These parasites attach themselves to the joints and gills of the crab causing breathing problems (www.fao.org).

Diseases of Chinese mitten crabs caused by viruses and bacteria (source: www.fao.org): i) shiver disease caused by a combination of a retrovirus and a bacteria causing shivering legs; ii) vibriosis: disease caused by a bacteria leading to dying of the legs and abdomen; iii) shell ulcer disease caused by bacteria solving chitine, a component of the carapace; iv) black gill disease caused by a bacteria affecting the gills leading to breathing problems. Other organisms like worms, nematoden, molluscs (snails and mussels), crustacea (Harpacticoida and Amphipoda), water mites (Halacarida) and Chironomidae have also been found on Chinese mitten crabs, but are not specifically related to this species and/or seem to affect these species (Normant et al., 2007)."

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* would have low or no genetic impact on other species in the RA area. No related species of the same genus are in the RA area.

Background Information: No native crabs in the same genus, so *E. sinensis* is unable to hybridize with local species (Stewart and Lockhart, 2005, Goldsmit et al., 2014) https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species **(Score = 2, Certainty = 1)**

Score Rationale: Little to no reliable information is available to suggest *Eriocheir sinensis* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. The species has the capacity to alter foreshore areas over great areas and prey on other species, thus it may have the capacity of impacting at risk bird and fish species through competition or cascading effects. However, interactions and potential effects in the RA area are not known. Note that the effects in freshwater were not considered.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and the at risk fish and bird species are unlikely to be affected. Beluga whales (*Delphinapterus leucas*) (COSEWIC 2004) might benefit from *E. sinensis* invasion by preying upon the crabs. There may be potential to harm freshwater species at risk hundreds of km upstream from the coast. In Europe they can move up to 1200 km upriver from the sea (Peters 1933 cited in Herborg et al., 2003).

Adults may impact freshwater SARA or COSEWIC-listed spp.

Q16- Impact on fisheries **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Eriocheir sinensis* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. The species has the capacity to alter foreshore areas over great areas and prey on other species. These facts can impact fisheries in the RA area. However, the extent of effects is uncertain as these will mainly be limited to estuarine phase of life history as there is no feeding during the fully marine phase.

Background Information: Juvenile *E. sinensis* might affect settling success of blue mussels which are harvested for subsistence but the crabs are unlikely to affect these harvests as they are small when they enter freshwater and are not feeding when they re-enter estuaries to reproduce. They might interfere with freshwater fisheries by scavenging bait or netted fish, fouling or damaging fishing gear (Panning, 1939, Rudnick and Resh, 2002) or preying on salmonid eggs or larvae (Culver, 2005).

From NEMESIS: “A variety of negative impacts of *Eriocheir sinensis* on fisheries have been reported from Europe and California. Mitten crabs steal fish bait from anglers and commercial fishermen in both regions. In some areas of San Francisco Bay and in European waters, interference by crabs with bait has made fishing unpleasant or impossible (Chinese Mitten Crab Working Group 2003; Invasive Species Specialist Group 2006). Sport fisheries in San Francisco Bay have a total annual value of about $2 billion per year (Chinese Mitten Crab Working Group 2003), so fishing losses in parts of the Bay can have a substantial economic impact. In addition to bait-stealing, dense populations of Mitten Crabs also interfere with traps, nets and aquaculture ponds, by clogging and breaking them, and by eating trapped fish (Chinese Mitten Crab Working Group 2003; Invasive Species Specialist Group 2006). In California, they caused serious mortality in fish-salvage facilities, designed to divert fish from irrigation facilities in the Sacramento-San Joaquin Delta. At peak migration times, mortality of migrating fish in 1998 was 98-99%. Retrofitting of the facilities to prevent mitten crab entrapment was expensive but necessary (Rudnick et al., 2000); Chinese Mitten Crab Working Group 2003). Mitten Crabs can

also interfere with fisheries by competing for food and shelter with fished species, such as crayfish and shrimp in San Francisco Bay (Chinese Mitten Crab Working Group 2003), or potentially with crab fisheries.”
(https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=99058)

From Therriault et al. (2008a): "In systems they have invaded, a number of adverse environmental consequences have been attributed to mitten crabs. For example, commercial trawl-fishermen have been forced to abandon certain areas in San Francisco Bay, California, USA and in Dutch estuaries when mitten crab populations were large (Ingle, 1986, Veldhuizen and Stanish, 1999). The impact on shellfish fishery was deemed moderate, which agrees with reports of mitten crabs eating thin shelled bivalves in rivers, nevertheless since the adults are not feeding during their marine phase the impact on marine shellfish fisheries will be very limited (if they exist at all). Each of finfish fishery, shellfish aquaculture and fisheries were predicted to be negatively impacted, which was an expected estimate due to the lack of feeding in the marine adult phase of the species."

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eriocheir sinensis* is invasive elsewhere in the world. Solid evidence of establishment and impacts in invaded areas.

Background Information: The native range of *E. sinensis* is in the western Pacific, along the Chinese coast and in large tributary rivers (Fofonoff et al., 2003, Dittel and Epifanio, 2009). It has invaded and is well established in Europe, and is established in San Francisco Bay along the US Pacific coast. In each of these areas the crabs’ distribution can extend many hundreds of kilometres upstream in freshwater tributaries. *Eriocheir sinensis* has been introduced along the Atlantic coast and in the Great Lakes of North America but may not be established. Introductions along the Arctic coasts of Europe and Asia and along the Gulf Coast to the US have failed.

The International Union for the Conservation of Nature and Natural Resources has placed *E. sinensis* on its list of the 100 most invasive alien species in the world because it has led to extinctions among native invertebrates, modified habitats by its intensive burrowing activities, caused losses in fisheries and aquaculture by consuming bait and trapped fish, and damaged gear (Lowe et al., 2000).

***Paralithodes camtschaticus* (Tilesius, 1815)**

Phylum: Arthropoda
  Class: Malacostraca
    Order: Decapoda
      Family: Lithodidae

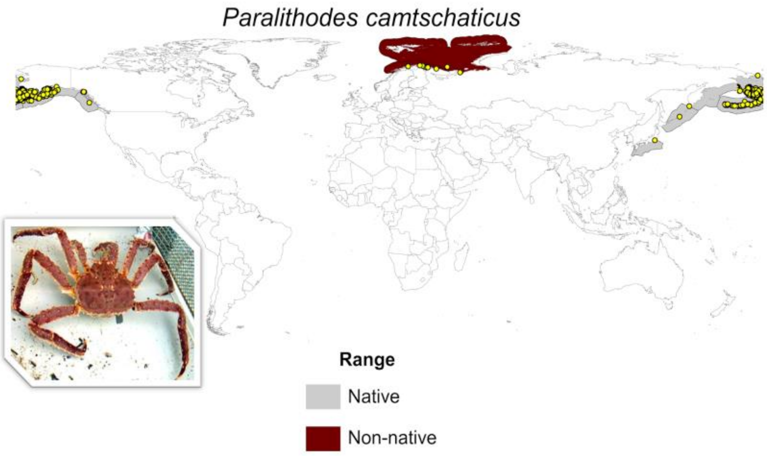

Figure 12: Ecoregions where Paralithodes camtschaticus is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Paralithodes camtschaticus occurrence points were obtained from GBIF.org (http://api.gbif.org/v0.9/occurrence/download/request/0009151-140429114108248.zip, 10 June 2014), Dvoretsky and Dvoretsky (2009b), Jørgensen and Nilssen (2011), Oug et al. (2011). Picture of P. camtschaticus mofidied from https://www.cabi.org/isc/datasheet/71549.

CMIST scores for *P. camtschaticus:*

Mean adjusted Likelihood of Invasion: 2.54

Mean adjusted Impact of Invasion: 2.43

Mean adjusted Overall CMIST score: 6.18

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Paralithodes camtschaticus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There have been no reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). Native to the Japan and Bering Seas, *P. camtschaticus* was intentionally introduced in the Barents Sea by Russian scientists to establish a new fishery in the 1960's. It is now spreading southward to Norwegian waters (Jørgensen, 2005) and recently has been found in the White Sea (Starikov et al., 2015).

Q2- Rate of introduction **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Paralithodes camtschaticus* could arrive infrequently in low numbers of individuals in the RA area. There are limited vectors for transport and it is not found in adjacent regions.

Background Information: For the red king crab, only the port of Tuktoyaktuk is connected to an international port where the species is present. It is very unlikely that this species would arrive through international ballast water in Canadian Arctic ports, and a null risk for domestic vessels

(Goldsmit et al., 2019). Future connections with Churchill port might be possible if Arctic Bridge is established (www.arcticbridge.com)

Q3- Survival (suitable habitat) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable habitat conditions for *Paralithodes camtschaticus.* There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: Paralithodes camtschaticus is found on soft bottoms over a range of 3 to ~370 m with size tending to increase with depth. It spawns in shallow rocky areas among seaweed but feeds on soft bottoms in deeper water (Brockerhoff and McLay, 2011).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Paralithodes camtschaticus*. There is a combination of viable temperature and salinity in most of the RA area. This is a species that resides in cold water regions that could easily find the environmental conditions in the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From CABI: “The red king crab is known to tolerate temperatures of -1.7 to 11 °C (Rodin, 1989) and this varies according to the life history stages. Little is known of the salinity tolerances: during ice periods it has been recorded to be present at 34 ppt (Hood et al., 1974), but during ice-free periods ranged from 22-24.5 ppt (Rusanowski et al., 1987).” (https://www.cabi.org/isc/datasheet/71549)

Models show that this species could find suitable habitat under present and future environmental conditions (Goldsmit et al., 2018).

Q5- Establishment (reproductive requirements) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are almost always available for *Paralithodes camtschaticus* in the RA area. Most stages of reproduction can withstand cold environments, within the range of the RA area. The species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From Jørgensen and Nilssen (2011): “Larvae develop in coastal zones and they settle in shallow waters (<20m) and they can settle on sponges, bryozoans and macroalgae (Marukawa, 1933). Larval survival is affected by the water temperature in which the egg carrying females had been kept. In the Kamchatka peninsula population adults assemble in shallow waters (10-15 m) in spring when temperature is approximately 2°C. Following reproduction in June-July, adults forage at around 50 m depth where the water is 2°C (Rodin, 1989).”

Lithodid crabs with an endotrophic mode of larval development have an outstanding capability to cope with temperatures that are typical of deep-sea and high latitudinal environments. Hypometabolism and optimization of all life cycle stages to low temperatures may be the physiological key to the success of this group in polar seas (Thatje et al., 2005). Larvae may tolerate water-column temperatures of -1.8 to 18°C at a salinity of 33.5 ppt (Nakanishi, 1985) but survive best at 5 to 10°C (Kurata 1960).

Q6- Establishment (natural control agents) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents are unlikely to affect population growth in the RA area for *Paralithodes camtschaticus*. There is considerable evidence to show this species is minimally affected by predators, except at the larval stage for which there seems to be a great deal of uncertainty, and climate being the most important factor regulating recruitment. Although there are known parasitic biocontrol agents for crabs, there are none specific to this species and uncertainty regarding native parasites in the RA area. There is considerable information on the species and its interactions.

Background Information: Parasites could act as a natural control agent in crabs, but as they have complex life cycles with intermediate hosts, the introduction of adult crabs together with their known parasites may be complicated. Native parasites in the new environment will not immediately attack a new host so crab numbers can explode in the absence of any restraints (Brockerhoff and McLay, 2011).

From Hansen (2015): “Some of the relevant natural enemies that can be used in marine biocontrol of crabs are parasitic castrators (Lafferty, 1993) and symbiotic egg predators (Goddard et al., 2005), but there is not a known parasite that is specific only for the red king crab.”

Declines in *P. camtschaticus* have been found to coincide with increased abundance of flatfish (mainly *Pleuronectes asper*), which prey on crab larvae and juveniles (Falk-Petersen et al., 2011; and references therein). In the Barents Sea, larvae are subject to predation by salmon (*Salmo salar*), saithe (*Pollachius virens*), and flatfish, including halibut (*Hippoglossus hippoglossus*) and flounder (including *Hippoglossoides platessoides*, *Microstomus kitt*, *Glyptocephalus cynoglossus*) (Falk-Petersen et al., 2011). Walleye pollock (*Theragra chalcogramma*) and sockeye salmon (*Oncorhynchus nerka*) have also been suggested as likely key predators of the larvae, however these relationships are uncertain and climate forcing has been speculated to be most important in determining larval recruitment (Falk-Petersen et al., 2011 and references therein).

From (Fuhrmann, 2016): “While invasive crustaceans may become a prey source for native fauna (De Rivera et al., 2005, Wong and Dowd, 2014), mega-decapods often have a low significance as prey for any organisms other than humans (Boudreau and Worm, 2012). Predation mortality for red king crabs estimated from Ecopath models was relatively low, and confirms the assumption that red king crabs experience low predation pressure from natural predators. This is most likely due to their size and spiny armour, and may be one of the major reasons for its success in colonizing new areas (see Lockwood et al., 2013). Predators on king crabs in its native areas are mostly pacific cod and large groundfish (reviewed in Falk-Petersen et al., 2011), feeding on crabs during or shortly after moulting. Predation by mammals may also occur on adults in the Pacific Ocean (Aydin et al., 2002). Some uncertainty around predators was given in our Ecopath models and more stomach samples from potential predators are necessary to improve diet input. Falk-Petersen et al. (2011) list salmon, saithe, a number of flatfish, skates and sculpins (Cottidae) as probable predators on crab larval stages and post-recruits in the Barents Sea. Ecopath models indicated that a small change in proportion of abundant predators, such as cod, may have a large impact on the crab population, despite the low proportions of crab in predator diets. Predation on juvenile crabs by sculpins (Aydin et al., 2007) has been observed in Porsangerfjord (H. K. Strand, IMR, pers. comm.) and may be significant in the inner area where sculpins are abundant. Cannibalism on small size classes was not documented in Porsangerfjord and is not seen as a major source of mortality in our study area. Cannibalism for red king crab has been documented from laboratory experiments (Borisov et al., 2007, Long et al., 2012) and Varangerfjord (Haugan, 2004; own observations). It has to be noted that cannibalism may be difficult to distinguish from consumption of own moults, common in crustaceans, leading to misconceptions about the importance of cannibalism in the field.”

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* has the capacity for a wide range of natural dispersal in the RA area.

Background Information: From Brockerhoff and McLay (2011): “Fecundity varies between 15,000 to nearly 500,000 eggs per female, depending on size (Jewett and Onuf, 1988). There are 4 planktonic stages plus a megalopa lasting about 2 months. Larvae settle at shallow depths (<20 m) and adults are found as deep as ~400 m on soft bottoms. Red king crabs perform seasonal migrations between shallow (spawning and mating areas in spring/summer) and deep waters (feeding areas in autumn/winter).”

The crab larvae may be pelagic for up to 60 days, dependent on environmental temperature, and may therefore be spread by currents (Sundet et al., 2000, Pedersen et al., 2006).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

*Score Rationale: S*ome reliable information suggests *Paralithodes camtschaticus* has the capacity for a wide range of anthropogenic dispersal in the RA area*.* There is good potential and supporting evidence for wide range dispersal based on shipping, through ballast in particular. Importance of other vectors such as biofouling is less clear and some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: Paralithodes camtschaticus was intentionally introduced to the Barents Sea by Russian scientists to establish a new fishery in the 1960’s (Jørgensen, 2005). Live crabs are imported for aquarium and restaurant trades (Ng, 1998), although in the Arctic region this trade does not take place. Ballast water transported by domestic coastal resupply and other boat traffic could spread *P. camtschaticus* larvae from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland 2009; Chan et al. 2012). Chan (2012) suggests that there is considerable domestic “Arctic direct” ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* would have high impacts in many areas on population growth in the RA area*.* There is good evidence that this species eats a wide variety of prey and can impact their population growth.

Background Information: Paralithodes camtschaticus causes reduction in soft-bottom communities in a number of larger individuals (Haugan, 2004, Anisimova et al., 2005); it is a large generalist predator that can prey on 100 different species affecting the abundance of different organisms (invertebrates and algae). It can cause a decrease in biomass of sipunculids, echinoderms and bivalves (Anisimova et al., 2005), though slow moving benthic animals such as echinoderms and shellfish bear the impact of king crab predation (Jørgensen, 2005). It can eliminate up to 15% of coastal populations of sea urchin (Gudimov et al., 2003) and heavily impact the mud star, *Ctenodsicus crispatus*, a species which is targeted mostly by large- and medium-sized red king crabs (Fuhrmann, 2016).

Red King crabs are important predators of many commercially harvested echinoderms and mollusc species, and eat the eggs of capelin (*Mallotus villosus*) (Anisimova et al., 2005, Britayev et al., 2010, Falk-Petersen et al., 2011, Oug et al., 2011). Reduced benthic diversity and biomass have been recorded in invaded areas (Falk-Petersen et al., 2011).

Paralithodes camtschaticus can cause competition with fish such as haddock, plaice, wolffish and cod (Pavlova et al., 2004, Anisimova et al., 2005).

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* would have high impact in many areas on communities in the RA area. There is good evidence that this species eats a wide variety of prey and can heavily impact benthic communities.

Background Information: Paralithodes camtschaticus is a large generalist predator that competes with native predators and decreases the amount of prey available for other native species. It causes reductions in soft-bottom communities in a number of larger individuals and causes shifts to dominance of small individuals (Haugan, 2004, Anisimova et al., 2005). It can prey on 100 different species, affecting the abundance of different organisms (invertebrates and algae). It can cause a decrease in biomass of sipunculids, echinoderms and bivalves (Anisimova et al., 2005), though slow moving benthic animals such as echinoderms and shellfish bear the impact of king crab predation (Jørgensen, 2005). Crab omnivory distributes predation pressure among various groups of organisms and prevents elimination of particular species or taxa (Britayev et al., 2010). *Paralithodes camtschaticus* can eliminate up to 15% of the coastal populations of sea urchin (Gudimov et al., 2003) and heavily impact the mud star, *Ctenodsicus crispatus*, a species which is targeted mostly by large- and medium-sized red king crabs (Fuhrmann, 2016). Reduced benthic diversity and biomass have been recorded in invaded areas (Falk-Petersen et al., 2011).

Paralithodes camtschaticus can cause competition with fish such as haddock, plaice, wolffish and cod (Pavlova et al., 2004, Anisimova et al., 2005).

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* would have high impacts in many areas on habitat in the RA area. There are well documented habitat effects from digging behavior which disrupts sediments and associated infaunal communities.

Background Information: These crabs are physical structures themselves and may represent new habitats that could allow increased biodiversity (Falk-Petersen et al., 2011). In the Barents Sea 13 different species were found as fouling on crabs carapace: barnacles and blue mussels were the most common, as well as polychaetes and amphipods. Epibionts grow faster on crabs than on fixed structures (Dvoretsky and Dvoretsky, 2009a). In general, large crabs, such as snow crab and red king crab, may impact sediment structure due to their foraging activities (Boudreau and Worm, 2012). The king crab clearly has the potential to reduce biodiversity and alter habitats (Falk-Petersen et al., 2011).

From Boudreau and Worm (2012): “As king crabs re-establish, they may impact the rich fauna of large amphipods and isopods found in this (Antarctic) region and could further affect benthic communities by consuming echinoderms and crustaceans (Thatje et al., 2005) and altering sediments through locomotive and feeding activities (Smith et al., 2011).”

From Smith et al. (2011): “Our data indicate that this king crab (*Neolithodes yaldwyni*) has major impacts on sediment texture, bioturbation and diversity of epibenthos. Numerous king crabs were observed walking across the Palmer Deep seafloor, creating puncture marks with leg tips and gashes in the sediment as leg tips were dragged through the sediment. Puncture marks from leg tips were approximately 1 cm in diameter and gashes were up to approximately 1 cm wide, approximately 0.5 cm deep and up to 20 cm long. In addition, crabs were observed to forage by repeatedly probing the sediment to 3–4 cm depths with open chelipeds, closing the chelipeds while the tips were submerged in the sediment, and then scooping the captured sediment bolus into open mouthparts for processing. This probing and scooping activity created sediment puncture marks and clumps 2– 3 cm across.”

From Oug et al. (2011): “In the invaded areas, the crab may therefore be expected to affect the benthic ecosystems both by predation on resident species and by disturbing the sedimentary environment from its digging activity. The infaunal species which increased in abundance have functional traits which may allow them to escape crab predation, such as small size (*Galathowenia*, *Myriochele*), by living in tough tubes (*Maldane*), or by digging deep into the sediment (*Thyasira*). It may, therefore, be suggested that removal of organisms that are important for sediment reworking and bio-irrigation leads to a degraded habitat quality. On the other hand, the crab may enhance particle mixing and oxygenation of surface sediments by its activity, such as walking, digging and scooping behavior. This is likely to be the case in Kobbholmfjorden, where there was a well-mixed surface layer and much evidence of crab tracks."

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* would have high impact in many areas on ecosystem function in the RA area. There is good evidence that this species has substantial ecosystem-level impacts.

Background Information: Paralithodes camtschaticus feed on epibenthic organisms that play an important role in the functioning of benthic systems (Rzhavsky et al., 2006). Since it is a generalist predator, it can disrupt food-web structure by competition with native predators and decrease the amount of prey available for other native species. It is known to compete with fish such as haddock, plaice, wolffish and cod (Pavlova et al., 2004, Anisimova et al., 2005). It can modify community structure by decreasing biomass of sipunculids, echinoderms and bivalves (Anisimova et al., 2005). Slow moving benthic animals such as echinoderms and shellfish can particularly be impacted by red king crab predation (Jørgensen, 2005). Dramatic impacts observed indicate that key ecosystems functions are affected (Oug et al., 2011) and that the ecosystem may lose resilience, which increases the likelihood of the ecosystem entering an alternative state (Falk-Petersen and Armstrong, 2013).

Fuhrmann (2016) combined field studies with ecosystem modeling (Ecopath) to show that these crabs have major impacts on ecosystem functioning: "Trophic niches of red king crab in Porsangerfjord overlapped with a few other native benthic predatory invertebrates such as other decapod crabs, sea stars and predatory snails, and bird groups, while most fish fed at higher trophic levels and/or used pelagic pathways. The red king crab had an important top-down role in the ecosystem, with a high overall impact. The crab itself was of low significance for other predators, impacting other species groups mostly through direct predation. EwE modelling showed that impacts were high on large, long-lived benthic invertebrates, often serving as prey and competitors at the same time. Predation by the red king crab on different trophic levels raised system omnivory, with unknown consequences for stability and resilience. Predation by the crabs possibly lead to a change in size structure in the benthic compartment, raising P/B ratios and leading to a faster turnover in biomasses. Increasing densities of red king crabs may also mediate regrowth of macroalgae by predation on sea urchins in some areas of the fjord, but this needs further investigation."

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Paralithodes camtschaticus* would have low or no impact associated with diseases, parasites or travelers in the RA area. The main vector of introduction of *P. camtschaticus* is ballast water containing the larval stage, hence all parasites and diseases that can be transported by adult crabs would be very low. However, there may be a possibility of expansion into the RA area by movement of adults (with associated parasites and diseases) established in other regions of the Arctic (e.g. Alaska) which would add some uncertainty as to whether risk could be higher.

Background Information: In trypanosome infections in Atlantic cod, crab carapace is a favoured substratum for the leech *Johanssonia arctica*, a vector for this parasite (Hemmingsen et al., 2005). The amphipod *Ischyrocerus commensalis* is an intermediate host of the trematode

Podocotyle atomon, a parasite of fish. It can influence fish parasitization patterns (Uspenskaya, 1963) and can have at least 43 associated epifaunal species (Dvoretsky and Dvoretsky, 2009a).

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* would have low or no genetic impact on other species in the RA area. No related species of the same genus are in the RA area.

Background Information: There are no native crabs in the same genus, so *P. camtschaticus* is unable to hybridize with local species (https://obis.org/). The NOBANIS fact sheet (https://www.nobanis.org/globalassets/speciesinfo/p/paralithodes-camtschatica/paralithodes_camtschaticus.pdf) states that there are no genetic effects associated with this species in the Barents Sea region.

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Paralithodes camtschaticus* would have high impacts in few areas or moderate impacts in many areas on at-risk or depleted species in the RA area. There are some benthic species at risk that could be impacted given the demonstrated effects of this crab species on benthic ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and the at-risk and bird species are unlikely to be affected. There are several at risk or special concern species of fish that could be affected through competition for prey – these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor* and the thorny skate, *Amblyraja radiata*. Walrus populations of special concern (COSEWIC 2006) may also be impacted as they feed on benthic invertebrates so may be impacted through competition for food resources or disturbance of benthic habitats by the red king crab. Beluga whales (*Delphinapterus leucas*) (COSEWIC 2004) (Species status and names from DFO 2020, https://www.dfo-mpo.gc.ca/species-especes/sara-lep/identify-eng.html) and perhaps other marine mammal species might benefit from a *P. camtschaticus* invasion by preying upon the crabs, which could reduce their other prey through direct or indirect competition.

Q16- Impact on fisheries (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Paralithodes camtschaticus* would have high impacts in many areas on fished species in the RA area. There are well demonstrated impacts on abundances of harvested species elsewhere and high potential for impacts in the RA area. There have been some mixed findings on *P. camtschaticus*' effects on fish species, though these may have been confounded by concurrent shifts in environmental conditions that also could have affected species abundances/composition.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries along the Nunavik coast; blue mussel (*Mytilus edulis*), green sea urchin (*Strongylocentrotus droebachiensis*), brown sea cucumber (*Cucumaria japonica*) and six-rayed starfish (*Leptasterias polaris*) have been harvested by commercial test fisheries in the Belcher Islands (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Commercial fisheries for shrimp are found in Hudson Strait and may be impacted by red king crab through food web/ecosystem changes (DFO 2018). Red king crabs are important predators of many commercially harvested echinoderm and mollusc species, and eat the eggs of capelin (*Mallotus villosus*) (Anisimova et al., 2005, Britayev et al., 2010, Falk-Petersen et al., 2011, Oug et al., 2011). These crabs could become bycatch (Brockerhoff and McLay, 2011) of coastal commercial and subsistence gillnet fisheries for anadromous fishes, particularly Arctic charr (*Salvelinus alpinus*) (Stewart and Lockhart, 2005), and cause damage to the fish and nets. Damage to shellfish populations in the Hudson Bay LME could harm harvested populations of bearded seal (*Erignathus barbatus*), Atlantic walrus (*Odobenus rosmarus*), and marine birds such as Hudson Bay eider (*Somateria mollissima*) that forage on benthic molluscs and other invertebrates. Some seals and whales (e.g., belugas and narwhals) may offset these impacts by eating these large crabs instead.

From Fuhrmann (2016): "Trophic niches of red king crab in Porsangerfjord overlapped with a few other native benthic predatory invertebrates such as other decapod crabs, sea stars and predatory snails, and bird groups, while most fish fed at higher trophic levels and/or used pelagic pathways. The red king crab had an important top-down role in the ecosystem, with a high overall impact. The crab itself was of low significance for other predators, impacting other species groups mostly through direct predation. EwE modelling showed that impacts were high on large, long-lived benthic invertebrates, often serving as prey and competitors at the same time. Predation by the red king crab on different trophic levels raised system omnivory, with unknown consequences for stability and resilience. Predation by the crabs possibly lead to a change in size structure in the benthic compartment, raising P/B ratios and leading to a faster turnover in biomasses. Increasing densities of red king crabs may also mediate regrowth of macroalgae by predation on sea urchins in some areas of the fjord, but this needs further investigation."

Pedersen et al. (2018) suggest that red king crabs may have cascading effects on multiple pathways, ultimately affecting harvested species, including fishes, benthic invertebrates, and birds.

Q17- Past invasion history **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Paralithodes camtschaticus* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Native from Okhotsk and Japan Seas, the Bering Sea and North Pacific Ocean, it was intentionally introduced in the Barents Sea by Russian scientists to establish a new fishery in 1960's. It is now spreading southward to Norwegian waters, (Jørgensen, 2005) and recently has been found in the White Sea (Starikov et al., 2015).

ZOOBENTHOS – MOLLUSCS

Littorina littorea (Linnaeus, 1758)

Phylum: Mollusca

Class: Gastropoda

Order: Littorinimorpha

Family: Littorinidae

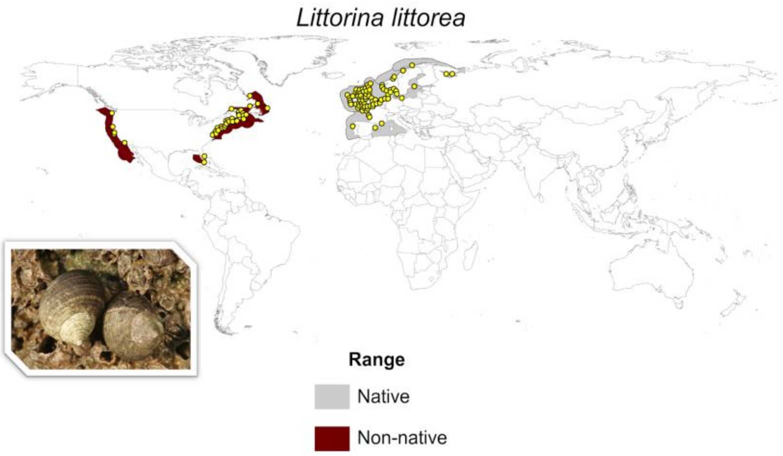

Figure 13: Ecoregions where Littorina littorea is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Littorina littorea occurrence points were obtained from OBIS (https://obis.org/) and GBIF.org (http://api.gbif.org/v0.9/occurrence/download/request/0009105-140429114108248.zip, 10 June 2014). Picture of L. littorea modified from https://invasions.si.edu/nemesis/species_summary/70419.

CMIST scores for *L. littorea:*

Mean adjusted Likelihood of Invasion: 2.36

Mean adjusted Impact of Invasion: 2.54

Mean adjusted Overall CMIST score: 5.99

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Littorina littorea* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There were no reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Goldsmit, 2016). In Atlantic Canada *L. littorea* has been established in the Gulf of St Lawrence since 1840, as well as in Nova Scotia and the Bay of Fundy (DFO 2012; Vercaemer et al., 2012). It has been periodically introduced along the Pacific coast north to Vancouver, BC, and has been collected but not yet established in BC (Harley et al., 2013). Specimens of *L. littorea* in the Canadian Museum of Nature are identified as this species for Hudson Bay from the 1960s, though confirmation of species is not possible because specimens were too small (Jean Marc Gagnon, CMN, pers. comm.). In McCann et al. (1981), *L. littorrea* is listed as present in Baffin Bay (information taken from Den Beste and McCart (1978)) - though these are highly probable to be misidentifications with native species *L. saxatilis.*

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Littorina littorea* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Ballast water is the most likely transport vector for introduction of *L. littorea* into the HB LME. Propagule pressure is unknown but these periwinkles are present at ports that are directly connected to Churchill and Deception Bay by ships travelling in ballast (Chan et al., 2012). Churchill had the highest annual likelihood of arrival per international vessel when compared to other ports. Nevertheless, these likelihoods can be considered low for all years (2005 to 2014) since the maximum likelihood was not high (Goldsmit et al., 2019).

Strong currents that flow east through southern Hudson Strait and then south along the Labrador coast should limit larval dispersal from Newfoundland into the Hudson Bay LME. Distance from existing *L. littorea* populations will limit transport by small boats and they are unlikely to be imported with seafood products or bait.

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Littorina littorea*. Reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers supports this classification.

Background Information: Littorina littorea are widely distributed on rocky coasts, in all except the most exposed areas, from the upper shore into the sublittoral zone (Fofonoff et al., 2003, Carlson et al., 2006) (https://www.cabi.org/isc/datasheet/76460). In sheltered conditions they also inhabit sandy or muddy habitats, such as estuaries and mud-flats. It is also found on manmade structures such as pilings in harbours.

Q4- Survival (suitable environment) **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Littorina littorea*. It can live in very cold environments and withstand freezing temperatures, and modelling results show that almost all Hudson Complex is suitable under current environmental conditions. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Littorina littorea can survive in a wide range of air (-13 to 35°C) and water (0 to 28°C) temperatures, and is fairly tolerant of brackish water (10-40 PSU; euryhaline) (https://www.cabi.org/isc/datasheet/76460). Its physiology makes it unusually well-adapted to being transported and established outside its native range. On cold coasts *L. littorea* typically moves from intertidal to subtidal areas. Almost all Hudson Complex is suitable under current environmental conditions according to habitat suitability modelling (Goldsmit et al., 2018).

Temperature tolerance can vary across the species' geographical range in response to acclimation (phenotypic) and adaptation (genotypic), but *L. littorea* is eurythermal at all sites (https://www.cabi.org/isc/datasheet/76460). It is generally found in water between 8 and 22°C. Golikov and Scarlato (1973) (cited in Christian et al., 2010) reported the summer and winter temperatures for the northern boundary of the species' distributional area were 6 and < 0°C, respectively, and for the southern boundary were 20 and 16°C, respectively, with optimum temperature ranges for inhabitance of 6 to 16°C. However, both freezing and heat tolerance of *L. littorea* decrease at lower salinities (~15 PSU) (Murphy, 1979, Clarke et al., 2000). Modelling results show that there would be a high proportion of the area that could be suitable for the species (Goldsmit et al., 2018).

Q5- Establishment (reproductive requirements) **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests reproductive requirements are almost always available for *Littorina littorea* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >= 2 °- 4 °C (minimum reproductive temperature) and the known wide salinity tolerance of this species, resulted in selecting the majority of the RA area as suitable for reproduction. There is good experimental and field data on reproductive requirements and environmental layers to evaluate the extent of the RA area that would have suitable conditions for reproduction.

Background Information: Reproduction is seasonal, usually including parts of the winter and spring (https://www.cabi.org/isc/datasheet/76460). Saucer-shaped capsules containing the zygotes are typically spawned at night on full and new moon spring tides during the spawning period. Water temperature controls the onset and duration of spawning. In New Brunswick, Chase and Thomas (1995) found that gonad maturation occurred from January to April and copulation occurred in late April to early May, while water temperatures were about 5 to 6 °C. In the lab spawning occurred at temperatures from 2 to 12°C, with maximum spawning at water temperatures of 7.3 to 9.6°C, and an optimum range of about 5 to 11°C. Colder temperatures may delay spawning and a slower increase in temperature can extend the spawning period to at least 24 weeks. Golikov and Scarlato (1973) (cited in Christian et al., 2010) reported an optimum temperature range for spawning of 4 to 16 °C. More information is needed on rearing success at low temperatures.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Littorina littorea*. Several studies show the ubiquity of predators on this species as well as parasites with appropriate hosts that could occur in the RA area and slow population growth. For example, gulls and wading birds - which are in the study area. Most information, although extensive, is general and extrapolated to species in the RA area.

Background Information: Littorina littorea are prey to sea stars (Harley et al., 2013), crabs, birds, whelks and other species. Predation by crabs and other species limits their subtidal abundance (Perez et al., 2009). Shell borers, both sponges (*Cliona* sp.) (Stefaniak et al., 2005) and polychaetes (*Polydora ciliata*) (Buschbaum et al., 2007), weaken shells and make *L. littorea*, particularly larger individuals, more susceptible to predation by crabs. The extent to which predation might control periwinkle populations in the HB LME is unknown. *Polydora* spp. and a variety of starfish are present in the region (Stewart and Lockhart, 2005).

From CABI: "Trematode parasites of *L. littorea* reduce reproductive success and survival by feeding on the female and male reproductive organs and the digestive tract (Fretter and Graham, 1962) and also reduce grazing rates of the snails (Wood et al., 2007). Blakeslee and Byers (2008) found 11 trematode species associated with European *L. littorea* and 5 species associated with North American *L. littorea* (e.g., *Cryptocotyle lingua* [most common in North America]; *Cercaria parvicaudata*, *Renicolaroscovita*, *Microphallus similis*). Trematode parasites of *L. littorea* are extensively studied throughout the snail's native and invasive ranges (e.g., Fretter and Graham, 1962, Kristoffersen, 1991, Blakeslee and Byers, 2008, Byers et al., 2008). *Littorina littorea* is infected by trematodes by ingesting bird faeces as it grazes in the intertidal zone, and forms one part of the infective life history (birds-snails-fishes) of the trematodes. *Cryptocotyle lingua*, the most common parasite in North American invasive populations, most likely present in RA given high densities of shorebirds including gulls which are a known final host; potential vectors for movement of *L. littorea* with this parasite exist via shipping." (https://www.cabi.org/isc/datasheet/76460)

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Littorina littorea* has the capacity for a wide range of natural dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal based on their extended planktonic larval stage.

Background Information: Littorina littorea are broadcast spawners that spawn annually from April to June (https://www.cabi.org/isc/datasheet/76460 and references therein). The planktotrophic larvae spend 2 to 7 weeks feeding in the water column before metamorphosis and settling. This period is shorter in warm and longer in cold water, increasing the species' potential dispersal in cold waters. Larvae of the *Littorina* spp. native to the HB LME are not planktotrophic (Panova et al., 2014).

From CABI: "This species disperses well because of its planktonic larva. The period that the larva (trochophore through veliger) spends in the plankton [stage] before metamorphosis and settlement is about 4-7 weeks (Thorson, 1946, Fretter and Graham, 1962, Williams, 1964)." (https://www.cabi.org/isc/datasheet/76460)

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Littorina littorea* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is solid support for wide range dispersal through ballast, and possibly biofouling, though the latter is more uncertain. There is also uncertainty as to the degree of ballast currently being uptaken, transported and released within the HBC.

Background Information: From CABI: "It is highly likely that *L. littorea* was introduced to North America from the intertidal or shallow subtidal zones of Great Britain and Ireland with rocks/cobbles used for ship ballast; ballast was dumped into the harbour and onshore in Pictou, Nova Scotia (and in other areas of Nova Scotia) in the late eighteenth and early nineteenth centuries (Brawley et al., 2009). In the twentieth century, local introductions of *L. littorea* to several bays on the Pacific U.S. shore are believed to have occurred as Atlantic shellfish like oysters and clams were transported for fisheries/aquaculture (Hanna, 1966, Carlton, 1969, Carlton, 1992, Carlton, 2007). Transport of *L. littorea* is occurring currently in bunches of *Ascophyllum nodosum*, because this brown seaweed is commonly used as packing material for northwestern Atlantic seafoods (e.g., lobsters) and worms (i.e., bait worms for fishermen) that are sold and shipped live to distant areas (Carlton, 2007); San Francisco Estuary Partnership, 2009). The type of *A. nodosum* that is being shipped with bait worms from Maine is, in fact, nicknamed “worm-weed” in Maine, and *L. littorea* (and *L. saxatilis*) occur on this seaweed in its natural shore habitat.” (https://www.cabi.org/isc/datasheet/76460). There is no aquaculture in the region at present to transport the species with seaweed packing materials.

Given the long planktonic larval stage (see Q7), ballast water transported by domestic coastal resupply and other boat traffic, and possibly hull fouling, could spread *L. littorea* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread. Human transport of adult periwinkles is also possible. Goldsmit et al. (2019) examined potential for transport of the species in the eastern Arctic and demonstrated a high potential risk for movement of *L. littorea* to several ports in the RA through ballast water. Although biofouling has not been documented, gastropods generally have been shown to occur in protected areas such as sea chests (Frey et al., 2014).

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Littorina littorea* would have high impacts in many areas on population growth in the RA area. There is clear evidence of major impacts on various related and unrelated taxa and communities

Background Information: Littorina littorea compete with and may have displaced native North American *Littorina saxatilis* from portions of the mid and low intertidal zone (Yamada and

Mansour, 1987, Eastwood et al., 2007). Both *L. saxatilis* and *Littorina obtusata* are native to the HB LME. Whether *L. littorea* would displace them there is unknown.

Impacts of *L. littorea* on the population growth of species in intertidal and upper subtidal habitats may be modified by the scouring action of thick ice cover.

Brenchley and Carlton (1983), demonstrated the impacts of *Littorina littorea* in Cape Cod Bay (MA, USA) on populations of native snails.

From The Exotics Guide: "Reviewing the history of coastal invasions, Carlton (1992) concluded that "no introduced marine mollusk in North America has had a greater ecological impact" than *Littorina littorea*, which altered the diversity, abundance, and distribution, of many animal and plant species on rocky as well as soft bottom shores." (https://www.exoticsguide.org/node/220)

From CABI: "One of the most important demonstrations of the ecological effects of *L. littorea* in the northwestern Atlantic was the experimental demonstration of its ability to shift the composition of algae on the shore (Lubchenco, 1978). High intertidal pools are filled with *Ulva intestinalis* and similar species when *L. littorea* is absent, but become dominated by crustose algae and the tough red seaweed *Mastocarpus stellatus* when the snail is present (Lubchenco, 1978). The rest of the intertidal shore can be grazed down to bare (or crust covered) rock at high *L. littorea* densities. At moderate densities, *L. littorea* can facilitate macrophyte growth by removing epiphytes, including diatoms, after the macrophytes (e.g., *Fucus* spp., Lubchenco, 1978) have grown large enough to avoid being removed by grazing *L. littorea*). It is likely that shores of the northwestern Atlantic had different species compositions (quantitatively) prior to the introduction of *L. littorea* (e.g., Vadas and Elner, 1992), but recent studies of the interactions of *L. littorea* and the native *L. saxatilis* (Yamada and Mansour, 1987) show competition and lead to a contrasting viewpoint (Eastwood et al., 2007). Eastwood et al. (2007) suggest that the grazing niche occupied by *L. saxatilis* extended throughout the intertidal zone prior to introduction of *L. littorea* and the predatory green crab *C. maenas*. Of course, there were still native fishes, birds, and crabs that are littorinid predators, and *L. saxatilis* has a thinner shell and is more susceptible to predation than *L. littorea*. Thus, demonstration of the niche overlap of these two snails (e.g., Eastwood et al., 2007) may still reflect a greater effect of *L. littorea* introduction on *L. saxatilis* than a potential equivalence of the two species as herbivores in the northwestern Atlantic intertidal zone. The consequences of grazing by *L. littorea* are much less dramatic on European shores than in North America because of the presence of large, herbivorous limpets in Europe that are absent in North America (i.e., *Patella vulgata*); these limpets are such major grazers that they swamp out the effects of *L. littorea*, but *L. littorea* is still a locally important herbivore in its native range (Hawkins et al., 1992).

Littorea littorea can physically alter salt marsh habitats and cobbled beaches by affecting sediment accretion (Bertness, 1984). As *L. littorea* browse cobble surfaces, they remove ("bulldoze") sediment from these surfaces. Grazing by *L. littorea* also reduces the amount of physically benign habitat at low tide for small organisms by removing leafy and filamentous green, red, and brown algae, which are otherwise often found in mats or turfs on the shore. There are both facilitations and inhibitions on settlement of invertebrate larvae (e.g., barnacle cyprids) associated with clearing of intertidal substratum by grazing *L. littorea* (e.g., Bertness, 1984, Buschbaum, 2000).

Grazing by *L. littorea* quantitatively reduces recruitment of many benthic intertidal organisms; larger sessile organisms (e.g., rockweeds) size-escape grazing and then benefit from being cleaned by surficial grazing of *L. littorea* on their surfaces (Lubchenco, 1983, Vadas and Elner, 1992). Locally, the quantity of green algae (in particular) is markedly reduced by the presence of *L. littorea*, but the overall biodiversity (species richness) of a shore is rarely affected because there are microhabitat refuges where such species escape grazing.

L. littorea in North America partly displaced the mud snail Ilyanassa obsoleta from mudflats, which may have had effects on the composition of infauna of mudflats (Brenchley and Carlton, 1983). " (https://www.cabi.org/isc/datasheet/76460).

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Littorina littorea* would have high impact in many areas on communities in the RA area. Clear evidence of major impacts on various related and unrelated taxa and communities.

Background Information: Littorina littorea is typically herbivorous but functions ecologically as an omnivore. As it scrapes surfaces with its radula, many juvenile invertebrates (e.g., recently settled barnacle larvae) are also ingested. Locally, the quantity of green algae (in particular) can be markedly reduced by the presence of *L. littorea*, but the overall biodiversity (species richness) of a shore is rarely affected due to microhabitat refuges where such species escape grazing (https://www.cabi.org/isc/datasheet/76460). Grazing by *L. littorea* quantitatively reduces recruitment of many benthic intertidal organisms; larger sessile organisms (e.g., rockweeds) escape grazing and then benefit from being cleaned by grazing of *L. littorea* on their surfaces (Lubchenco, 1983).

Littorina littorea eats many eggs of the native mud snail *Ilyanassa obsoleta* in mid-intertidal habitats on the Atlantic coast (Brenchley, 1982). It has partly displaced the mudsnail from mudflats, which may have affected the composition of mudflat infauna (Brenchley and Carlton, 1983). *Littorina littorea* competes with and may have displaced native North American shore snail *L. saxatilis* from portions of the mid and low intertidal zone (Yamada and Mansour, 1987, Eastwood et al., 2007). In experimental studies, *L. littorea* depressed the species' growth rate (Yamada and Mansour, 1987); it also negatively affected growth of the native limpet *Tectura testudinalis* (Petraitis, 1989).

Impacts of *L. littorea* on intertidal and upper subtidal communities may be modified by ice scour.

From The Exotics Guide: "Reviewing the history of coastal invasions, Carlton (1992) concluded that "no introduced marine mollusk in North America has had a greater ecological impact" than *Littorina littorea*, which altered the diversity, abundance, and distribution, of many animal and plant species on rocky as well as soft bottom shores." (https://www.exoticsguide.org/node/220)

From CABI: "One of the most important demonstrations of the ecological effects of *L. littorea* in the northwestern Atlantic was the experimental demonstration of its ability to shift the composition of algae on the shore (Lubchenco, 1978). High intertidal pools are filled with *Ulva intestinalis* and similar species when *L. littorea* is absent, but become dominated by crustose algae and the tough red seaweed *Mastocarpus stellatus* when the snail is present (Lubchenco, 1978). The rest of the intertidal shore can be grazed down to bare (or crust covered) rock at high *L. littorea* densities. At moderate densities, *L. littorea* can facilitate macrophyte growth by removing epiphytes, including diatoms, after the macrophytes (e.g., *Fucus* spp., Lubchenco, 1978) have grown large enough to avoid being removed by grazing *L. littorea*). It is likely that shores of the northwestern Atlantic had different species compositions (quantitatively) prior to the introduction of *L. littorea* (e.g., Vadas and Elner, 1992), but recent studies of the interactions of *L. littorea* and the native *L. saxatilis* (Yamada and Mansour, 1987) show competition and lead to a contrasting viewpoint (Eastwood et al., 2007). Eastwood et al. (2007) suggest that the grazing niche occupied by *L. saxatilis* extended throughout the intertidal zone prior to introduction of *L. littorea* and the predatory green crab *C. maenas*. Of course, there were still native fishes, birds, and crabs that are littorinid predators, and *L. saxatilis* has a thinner shell and is more susceptible to predation than *L. littorea*. Thus, demonstration of the niche overlap of these two snails (e.g., Eastwood et al., 2007) may still reflect a greater effect of *L. littorea* introduction on *L. saxatilis* than a potential equivalence of the two species as herbivores in the northwestern Atlantic intertidal zone. The consequences of grazing by *L. littorea* are much less dramatic on European shores than in North America because of the presence of large, herbivorous limpets in Europe that are absent in North America (i.e., *Patella vulgata*); these limpets are such major grazers that they swamp out the effects of *L. littorea*, but *L. littorea* is still a locally important herbivore in its native range (Hawkins et al., 1992).

Littorea littorea can physically alter salt marsh habitats and cobbled beaches by affecting sediment accretion (Bertness, 1984). As *L. littorea* browse cobble surfaces, they remove ("bulldoze") sediment from these surfaces. Grazing by *L. littorea* also reduces the amount of physically benign habitat at low tide for small organisms by removing leafy and filamentous green, red, and brown algae, which are otherwise often found in mats or turfs on the shore. There are both facilitations and inhibitions on settlement of invertebrate larvae (e.g., barnacle cyprids) associated with clearing of intertidal substratum by grazing *L. littorea* (e.g., Bertness, 1984, Buschbaum, 2000).

Grazing by *L. littorea* quantitatively reduces recruitment of many benthic intertidal organisms; larger sessile organisms (e.g., rockweeds) size-escape grazing and then benefit from being cleaned by surficial grazing of *L. littorea* on their surfaces (Lubchenco, 1983, Vadas and Elner, 1992). Locally, the quantity of green algae (in particular) is markedly reduced by the presence of *L. littorea*, but the overall biodiversity (species richness) of a shore is rarely affected because there are microhabitat refuges where such species escape grazing.

Of course, there were still native fishes, birds, and crabs that are littorinid predators, and *L. saxatilis* has a thinner shell and is more susceptible to predation than *L. littorea*. Thus, demonstration of the niche overlap of these two snails (e.g., Eastwood et al., 2007) may still reflect a greater effect of *L. littorea* introduction on *L. saxatilis* than a potential equivalence of the two species as herbivores in the northwestern Atlantic intertidal zone.

L. littorea in North America partly displaced the mud snail *Ilyanassa obsoleta* from mudflats, which may have had effects on the composition of infauna of mudflats (Brenchley and Carlton, 1983). As noted above, *L. littorea* competes with and may have displaced native North American *L. saxatilis* from portions of the mid and low intertidal zone (Yamada and Mansour, 1987, Eastwood et al., 2007)." (https://www.cabi.org/isc/datasheet/76460)

Q11- Impact on habitats **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Littorina littorea* would have high impact in many areas on habitat in the RA area. There are well documented changes in Habitat structure related to this species ((e.g. muddy to rocky habitats, replacement of algal components).

Background Information: Littorina littorea can reach intertidal abundances of over 100,000/m^2 on rocky shores, cobbled beaches, mudflats, and *Spartina marshes* (e.g., Lubchenco, 1978, Bertness, 1984, Buschbaum, 2000, Carlson et al., 2006, Tyrrell et al., 2008). It can physically alter salt marsh habitats and cobbled beaches by affecting sediment accretion (Bertness, 1984). As *L. littorea* browse cobble surfaces, they remove ("bulldoze") sediment from these surfaces. Their grazing also reduces the amount of physically benign habitat at low tide for small organisms by removing leafy and filamentous green, red, and brown algae, which are otherwise often found in mats or turfs on the shore. By clearing intertidal substratum they can both facilitate and inhibit settlement of invertebrate larvae (e.g., barnacle cyprids) (e.g., Bertness, 1984, Buschbaum, 2000) (https://www.cabi.org/isc/datasheet/76460).

Impacts of *L. littorea* on intertidal and upper subtidal habitats may be modified by the scouring action of thick ice cover.

From The Exotics Guide: “Reviewing the history of coastal invasions, Carlton (1992) concluded that “no introduced marine mollusk in North America has had a greater ecological impact” than *Littorina littorea*, which altered the diversity, abundance, and distribution, of many animal and plant species on rocky as well as soft bottom shores.” (https://www.exoticsguide.org/node/220)

From CABI: “One of the most important demonstrations of the ecological effects of *L. littorea* in the northwestern Atlantic was the experimental demonstration of its ability to shift the composition of algae on the shore (Lubchenco, 1978). High intertidal pools are filled with *Ulva intestinalis* and similar species when *L. littorea* is absent, but become dominated by crustose algae and the tough red seaweed *Mastocarpus stellatus* when the snail is present (Lubchenco, 1978). The rest of the intertidal shore can be grazed down to bare (or crust covered) rock at high *L. littorea* densities. At moderate densities, *L. littorea* can facilitate macrophyte growth by removing epiphytes, including diatoms, after the macrophytes (e.g., *Fucus* spp., Lubchenco, 1978) have grown large enough to avoid being removed by grazing *L. littorea*). It is likely that shores of the northwestern Atlantic had different species compositions (quantitatively) prior to the introduction of *L. littorea* (e.g., Vadas and Elner, 1992), but recent studies of the interactions of *L. littorea* and the native *L. saxatilis* (Yamada and Mansour, 1987) show competition and lead to a contrasting viewpoint (Eastwood et al., 2007). Eastwood et al. (2007) suggest that the grazing niche occupied by *L. saxatilis* extended throughout the intertidal zone prior to introduction of *L. littorea* and the predatory green crab *C. maenas*…[ ]… The consequences of grazing by *L. littorea* are much less dramatic on European shores than in North America because of the presence of large, herbivorous limpets in Europe that are absent in North America (i.e., *Patella vulgata*); these limpets are such major grazers that they swamp out the effects of *L. littorea*, but *L. littorea* is still a locally important herbivore in its native range (Hawkins et al., 1992).

Littorea littorea can physically alter salt marsh habitats and cobbled beaches by affecting sediment accretion (Bertness, 1984). As *L. littorea* browse cobble surfaces, they remove (“bulldoze”) sediment from these surfaces. Grazing by *L. littorea* also reduces the amount of physically benign habitat at low tide for small organisms by removing leafy and filamentous green, red, and brown algae, which are otherwise often found in mats or turfs on the shore. There are both facilitations and inhibitions on settlement of invertebrate larvae (e.g., barnacle cyprids) associated with clearing of intertidal substratum by grazing *L. littorea* (e.g., Bertness, 1984, Buschbaum, 2000).

Grazing by *L. littorea* quantitatively reduces recruitment of many benthic intertidal organisms; larger sessile organisms (e.g., rockweeds) size-escape grazing and then benefit from being cleaned by surficial grazing of *L. littorea* on their surfaces (Lubchenco, 1983, Vadas and Elner, 1992). Locally, the quantity of green algae (in particular) is markedly reduced by the presence of *L. littorea*, but the overall biodiversity (species richness) of a shore is rarely affected because there are microhabitat refuges where such species escape grazing.” (https://www.cabi.org/isc/datasheet/76460)

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Littorina littorea* would have high impact in many areas on ecosystem function in the RA area. There are well documented changes in ecosystem functioning (e.g. in muddy to rocky habitats, replacement of algal components) related to this species.

Background Information: No introduced marine mollusk in North America has had a greater ecological impact than *L. littorea*, which colonized the Atlantic coast from Nova Scotia to New Jersey (Carlton, 1992). Since colonizing the region, from 1860 to 1890, it has altered the diversity, abundance, and distribution of many animal and plant species on rocky as well as soft bottom shores. *Littorina littorea* is now the most common marine snail along this coast. This periwinkle can fundamentally change intertidal ecosystems via grazing activities, altering the distribution and abundance of algae on rocky shores and converting soft-sediment habitats to hard substrates. Its ability to shift the composition of algae on the shore has been demonstrated experimentally (Lubchenco, 1978).

From The Exotics Guide: “Reviewing the history of coastal invasions, Carlton (1992) concluded that “no introduced marine mollusk in North America has had a greater ecological impact” than *Littorina littorea*, which altered the diversity, abundance, and distribution, of many animal and plant species on rocky as well as soft bottom shores.” (https://www.exoticsguide.org/node/220)

From CABI: “One of the most important demonstrations of the ecological effects of *L. littorea* in the northwestern Atlantic was the experimental demonstration of its ability to shift the composition of algae on the shore (Lubchenco, 1978). High intertidal pools are filled with *Ulva intestinalis* and similar species when *L. littorea* is absent, but become dominated by crustose algae and the tough red seaweed *Mastocarpus stellatus* when the snail is present (Lubchenco, 1978). The rest of the intertidal shore can be grazed down to bare (or crust covered) rock at high *L. littorea* densities. At moderate densities, *L. littorea* can facilitate macrophyte growth by removing epiphytes, including diatoms, after the macrophytes (e.g., *Fucus* spp., Lubchenco, 1978) have grown large enough to avoid being removed by grazing *L. littorea*). It is likely that shores of the northwestern Atlantic had different species compositions (quantitatively) prior to the introduction of *L. littorea* (e.g., Vadas and Elner, 1992), but recent studies of the interactions of *L. littorea* and the native *L. saxatilis* (Yamada and Mansour, 1987) show competition and lead to a contrasting viewpoint (Eastwood et al., 2007). Eastwood et al. (2007) suggest that the grazing niche occupied by *L. saxatilis* extended throughout the intertidal zone prior to introduction of *L. littorea* and the predatory green crab *C. maenas*. Of course, there were still native fishes, birds, and crabs that are littorinid predators, and *L. saxatilis* has a thinner shell and is more susceptible to predation than *L. littorea*. Thus, demonstration of the niche overlap of these two snails (e.g., Eastwood et al., 2007) may still reflect a greater effect of *L. littorea* introduction on *L. saxatilis* than a potential equivalence of the two species as herbivores in the northwestern Atlantic intertidal zone. The consequences of grazing by *L. littorea* are much less dramatic on European shores than in North America because of the presence of large, herbivorous limpets in Europe that are absent in North America (i.e., *Patella vulgata*); these limpets are such major grazers that they swamp out the effects of *L. littorea*, but *L. littorea* is still a locally important herbivore in its native range (Hawkins et al., 1992).

Littorea littorea can physically alter salt marsh habitats and cobbled beaches by affecting sediment accretion (Bertness, 1984). As *L. littorea* browse cobble surfaces, they remove (“bulldoze”) sediment from these surfaces. Grazing by *L. littorea* also reduces the amount of physically benign habitat at low tide for small organisms by removing leafy and filamentous green, red, and brown algae, which are otherwise often found in mats or turfs on the shore. There are both facilitations and inhibitions on settlement of invertebrate larvae (e.g., barnacle cyprids) associated with clearing of intertidal substratum by grazing *L. littorea* (e.g., Bertness, 1984, Buschbaum, 2000).

Grazing by *L. littorea* quantitatively reduces recruitment of many benthic intertidal organisms; larger sessile organisms (e.g., rockweeds) size-escape grazing and then benefit from being cleaned by surficial grazing of *L. littorea* on their surfaces (Lubchenco, 1983, Vadas and Elner, 1992). Locally, the quantity of green algae (in particular) is markedly reduced by the presence of *L. littorea*, but the overall biodiversity (species richness) of a shore is rarely affected because there are microhabitat refuges where such species escape grazing.

L. littorea in North America partly displaced the mud snail Ilyanassa obsoleta from mudflats, which may have had effects on the composition of infauna of mudflats (Brenchley and Carlton, 1983). As noted above, *L. littorea* competes with and may have displaced native North

American *L. saxatilis* from portions of the mid and low intertidal zone (Yamada and Mansour, 1987, Eastwood et al., 2007).” (https://www.cabi.org/isc/datasheet/76460).

Q13- Associated diseases, parasites, and travellers (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Littorina littorea* would have high impact in many areas associated with diseases, parasites or travelers in the RA area. There is much evidence showing impacts/importance of the species for parasite transmission.

Background Information: Littorina littorea is host to obligate and facultative endosymbionts from seven animal phyla, including Annelida, Arthropoda, Gnathostomulida, Nematoda, Nemertea, and Platyhelminthes, as well as ciliates and algae (Buckland-Nicks et al., 2013). Nikolaev et al. (2020) state that intertidal molluscan hosts in the White Sea are infected with trematodes year-round. *Littorina littorea* was likely the vector for introducing the trematode *Cryptocotyle lingua* from Europe to North America (Blakeslee and Byers, 2008, Harley et al., 2013). This parasite requires snails, fish, and birds or mammals to complete its life cycle. Periwinkles eat the eggs--likely with bird feces while grazing, which develop into rediae in the host before developing into swimming cercariae. Heavy infections damage the periwinkle's digestive organs and reduce grazing and reproductive rates. The cercariae infect fish and develop into metacercariae, leaving unpleasant tissue discolouration and heavy infections sometimes kill the host (e.g., Atlantic herring *Clupea harengus*). The trematodes mature into adults in birds (e.g., *Larus argentatus*, *Gavia immer*) or mammals (e.g., dogs) that eat the fish, and then shed eggs back into the water (Daoust et al., 1998, Fofonoff et al., 2003). Terns (probably the common tern *Sterna sterna*) and dogs fed infected fish suffered localized intestinal damage (Willey and Stunkard, 1942; cited in Fofonoff et al. 2003). If introduced to the HB LME these trematodes are likely to parasitize the native *Littorina* spp. and infect a variety of species in three different trophic levels within the region.

Many of the species listed above are quite host-specific and would likely not affect other littorines. But they may infect the other ecosystem components (fish, gulls, etc...).

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Littorina littorea* would have low or no genetic impact on other species in the RA area. There are co-occurring native littorines of the same genus in the RA area, but no evidence of hybridization found in the literature.

Background Information: Two other *Littorina* species, *L. obtusata* and *L. saxatilis*, occur in the Hudson Bay LME (Stewart and Lockhart, 2005). These species are more closely related to each other than to *L. littorea* (Panova et al., 2014). They reproduce throughout the year and have non-pelagic larvae, whereas *L. littorea* has pelagic (planktotrophic) larvae and a distinct breeding season (Erlandsson, 2002). These differences should limit reproductive overlap between the native species and *L. littorea*; no information was found to suggest they will hybridize.

Some littorine species can hybridize. From Warwick et al. (1990): *L. saxatilis* could be crossed with female *L. arcana*, but the reverse was not true; the female hybrid progeny could be backcrossed to male *L. saxatilis*.

Although there is some evidence of hybridization in *Littorina* spp. (García-Souto et al., 2018, Costa et al., 2020), there is no direct evidence of *L. Littorea* hybridizing with *L. saxatilis* or *L. obtusata* (perhaps related to clear differences in reproductive strategy – planktonic larvae for *L. littorea* and egg masses/brood pouch for *L. saxatilis* and *L. obtusata*) and overall differences in life histories/ecologies in this extremely well-studied group of marine gastropods (Rolán-Alvarez et al., 2015).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Littorina littorea* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. Since the species has the ability to alter foreshores over great areas, it may have the capacity to impact at risk bird species through cascading effects or acting as a vector for parasites. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and at-risk fish and mammal species are unlikely to be affected. But, the exposure of bird species at risk (www.cosewic.gc.ca) (ivory gull *Pagophila eburnea* (endangered) and Ross's gull *Rhodostethia rosea* (threatened)) to infections of *Cryptocotyle lingua* might increase if this parasite is introduced with periwinkles.

Q16- Impact on fisheries **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Littorina littorea* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. The species has the ability to alter foreshores over great areas, thus it may have the capacity of impacting at risk and fisheries species either directly or through cascading effects.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Commercially harvested anadromous Arctic charr (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals.

Q17- Past invasion history **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Littorina littorea* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Littorina littorea is native to Europe, from the White Sea to Gibraltar (Fofonoff et al., 2003). It is invasive in North America where it has established on the Atlantic coast from Lewes, Delaware to Red Bay, Labrador (Chapman et al., 2007, Blakeslee and Byers, 2008, Brawley et al., 2009; see also Wares et al. (2002)). However, the introductions along the Pacific coast of North America and in the Mediterranean Sea have so far failed to establish (Fofonoff et al., 2003). Individuals have been found in BC waters but their population structure and lack of recruits suggest they are not yet established (Harley et al., 2013).

Mya arenaria Linnaeus, 1758

Phylum: Mollusca
Class: Bivalvia
Order: Myida
Family: Myidae

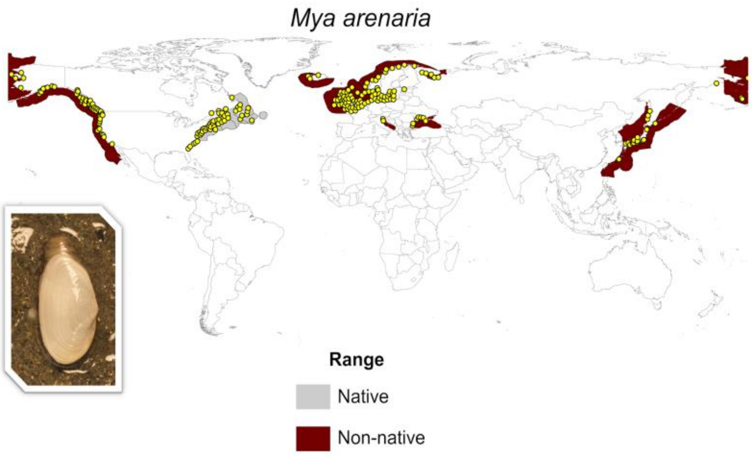

Figure 14: Ecoregions where Mya arenaria is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Mya arenaria occurrence points were obtained from OBIS (https://obis.org/) and GBIF.org (http://doi.org/10.15468/dl.kai1wm, 10 April 2017). Picture of M. arenaria modified from https://invasions.si.edu/nemesis/species_summary/-60.

CMIST scores for *M. arenaria:*

Mean adjusted Likelihood of Invasion: 2.31

Mean adjusted Impact of Invasion: 2.59

Mean adjusted Overall CMIST score: 5.99

Q1- Present status in the area (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Mya arenaria* has been observed and is not reported as established in the RA area. However, survey effort is low relative to the size of the RA area. Also there is some uncertainty regarding whether it has been found in the RA or not: According to Zhang et al. (2018) and references therein, historical reports of *M. arenaria* from the Arctic Ocean are erroneous and likely represent *Mya truncata ovata* Jensen, 1900, which is currently recognized as a junior synonym of *Mya pseudoarenaria*. However, they have identified *M. arenaria* in the Russian waters of the Barents Sea, a marginal sea of the Arctic Ocean, and it has also been reported further west near Forsøl, Norway.

Background Information: No reports have been found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). Although Lubinsky (1980), Bussières et al. (2008), and Roy and Gagnon (2016) reported isolated occurrences from Hudson Bay LME, these may have been misidentified. The native *Mya truncata* are widely distributed in the Hudson Bay LME and *M. pseudoarenaria* are present in southeastern Hudson Bay and James Bay. Both are native to the Canadian Arctic and the latter is very similar morphologically, making it difficult to differentiate from *M. arenaria*. That said, Chain et al. (2016) and Brown et al. (2016) identified *Mya arenaria* from metabarcoding of zooplankton samples collected in the Churchill area in 2011 suggesting the possible occurrence of the species there. *Mya arenaria* is indigenous on the Atlantic coast from South Carolina to southern Labrador (Lubinsky, 1980, Hicks and Ouellette, 2011).

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Mya arenaria* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Propagule pressure is unknown, but *M. arenaria* is present at ports directly connected to Churchill and Deception Bay by shipping. It could be entrained and transported by these vessels in ballast water, or on fouled hulls (Chan et al., 2012). Four ports of arrival received vessels with domestic ballast water originating from regions where *M. arenaria* was present. Among these, Deception Bay (years 2005, 2006, 2008, 2013 and 2014) and Churchill (year 2005) had the highest annual likelihood of arrival per vessel. Ten ports of arrival received vessels with international ballast water originating from regions where *M. arenaria* was present. Churchill had the highest annual likelihood of arrival per international vessel when compared to other ports. Nevertheless, these likelihoods can be considered low for all years (2005 to 2014) since the maximum likelihood was not high. For all other ports receiving international vessels, likelihoods of arrival were zero or close to zero since low quantities of ballast water were discharged or no discharge at all was associated with ballast water coming from places where this species is known to occur (Goldsmit et al., 2019).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Mya arenaria*. There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: Mya arenaria inhabits gravelly to muddy bottoms, from the mid-intertidal to about 100 m depth, though they are rare below 9-10 m (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60). *Mya arenaria* is capable of surviving and reproducing at lower salinities than many native Pacific species, where it is often the dominant (or only) marine bivalve in upper estuaries. It tends to occur higher in the intertidal zone than the native clams, so its invasion success may have been due to filling an unoccupied niche (Fofonoff et al., 2003). These attributes may not give it the same competitive advantage in Arctic waters where salinities tend to be lower, particularly during the spring freshet and ice melt, and where ice scour and exposure to cold create a very harsh intertidal zone.

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Mya arenaria*. It can tolerate low temperatures such as -2° C making most of the assessment area available, together with salinity tolerances. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Mya arenaria is tolerant of low salinity and quite large changes in salinity and temperature. Adults can tolerate salinities as low as 5 ppt, and temperatures from -2 to 28° C and can survive in an oxygen-free environment for up to 8 days (https://www.exoticsguide.org/mya_arenaria). Salinity tolerance of the softshell clam appears to be linked to water temperature. Generally, the preferred temperature range of this clam is 6 to 14 ºC, but it can withstand freezing for up to 7 weeks (Caddy et al., 1974; from Christian et al., 2010).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Mya arenaria* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >= 12 °C (ideal conditions for reproduction) and >=10 °C (ideal conditions for larval development) resulted in a moderate amount of habitat available in the RA area. The species requirements are well demonstrated and there are reliable environmental data layers for the RA area to show a match.

Background Information: Mya arenaria usually have separate sexes, but there is a low incidence of hermaphroditism*.* They usually spawn twice a year, in spring and fall, mostly in the southern part of its range (Connecticut, Rhode Island, but also in Oslofjord, Norway and southern England). They only spawn once a year further north (White Sea- Russia, Maine, New Brunswick, Ireland, Sweden, Wadden Sea, but also the Black Sea). Spawning usually occurs at 10 to 25ºC, but the temperature range is quite variable and they do not tolerate temperatures above 28ºC for prolonged periods (Sadykhova, 1979, Newell et al., 1986, Strasser, 1998, Cross et al., 2012; https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60). They have been shown to occur in spring at temperatures of 4-6 ºC in Massachusetts (Brousseau, 1978).

In laboratory cultures, the optimum temperature range for larval development was between about 17.7 to 23°C, although slow development took place as low as 10°C and optimum salinity was from about 16% to 32% (the highest value tested) (Stickney, 1964b), while growth of juveniles was estimated to cease at 3.7 °C (Stickney, 1964a) and feeding of juveniles ceased at 2.8 °C (Belding 1930 in Stickney, 1964a).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents could slow population growth in the RA area for *Mya arenaria*. There are closely related species in the RA area that could act as predators but no indication that these would severely limit population growth, particularly at the adult stage. There is substantial information on predators of this species.

Background Information: From NOBANIS: “Crabs (e.g. *Carcinus maenas* and *Callinectes sapidus*) and flat-fish (flounder and plaice) are the main predators on *M. arenaria*. Brown shrimp, *Crangon crangon* and gobies (*Pomatoschistus microps*) may also feed on small *M. arenaria* (Möller and Rosenberg, 1983, Günther, 1992). Large clams burrow so deep that only a few wading birds, such as curlew, have beaks long enough to catch them (Zwarts and Wanink, 1989). However, tips of siphons may still be nibbled off by surface feeding predators. Small clams are eaten by fishes, crabs, clam worms (Nereidae), moon snails (Naticidae), birds, etc. When clams reach ~60 mm in length, they are less vulnerable to predation (Newell et al., 1986).” (https://www.nobanis.org/globalassets/speciesinfo/m/mya-arenaria/mya-arenaria.pdf)

From Christian et al. (2010): “Nemerteans are important predators of newly settled larvae in marine soft bottom communities, especially in the intertidal areas (Ambrose Jr, 1991, Bourque et al., 2001a, Bourque et al., 2001b). Bourque et al. (2002) described the searching and feeding behaviors of a common Atlantic Canada nemertean, *Cerebratulus lacteus*, a known predator of *M. arenaria*.”

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mya arenaria* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for potential natural dispersal from larval transport.

Background Information: Larvae remain planktonic for 2 to 3 weeks prior to settlement and can be carried substantial distances from their hatching location (Christian et al., 2010). Spat may remain in a floating/crawling mode for 2 to 5 weeks, attaching to the substrate (eelgrass, filamentous algae, abiotic substrate) with byssal threads. As they grow larger, the spat burrows into the soft sediment, with burrowing depth increasing with age and size. Once these animals reach a shell length of approximately 5 mm, they are referred to as "seed clams". Juvenile seed clams may migrate shoreward as far as several hundred meters (Christian et al., 2010).

Smaller (1.3 mm) *M. arenaria* tended to disperse more than larger (3.7 mm) ones. Dispersal distances due to bedload transport are likely to be in centimeters per hour. Although these dispersal distances are small, such movements are likely to occur frequently due to tidal currents and, consequently, may have profound impacts on patterns of abundance and distribution (Jennings and Hunt, 2009).

Q8- Spread (potential anthropogenic dispersal) **(Score = 3, Certainty = 2)**

Score Rationale: Some reliable information suggests *Mya arenaria* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence of dispersal due to larval transport in ballast but some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: From The Exotics Guide: *Mya arenaria* were introduced with oysters on the west coast of North America in the late 1800s (Palacios et al., 2000). The spread to some sites may also have occurred accidentally through transplanting of oysters along the coast or with fresh introductions of oysters from the Atlantic. It is possible, though less likely, that *M. arenaria*'s appearance in some locations resulted from deliberate introductions from the Atlantic, as some authors have claimed was attempted or occurred, or from the transport of small clams in ship fouling. (https://www.exoticsguide.org/mya_arenaria)

Mya arenaria invaded new habitats by different modes: natural range expansion, intentional introductions, unintentional introductions as a ballast species and unintentional introduction as a byproduct of oyster transplants (Strasser, 1998). In the assessment region, the only way of introduction by anthropogenic mechanisms would be by ballast water. Ballast transported by domestic coastal resupply and other boat traffic, could spread *M. arenaria* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Mya arenaria* would have high impacts in many areas on population growth in the RA area. There are clear major effects on various populations which are well documented.

Background Information: From NEMESIS: "When abundant, *M. arenaria* is a significant herbivore in estuaries, because of its large size and powerful filtration, and its ability to survive in low salinities and wide tidal ranges, where large native bivalves are often rare.

Introduced populations of *M. arenaria* in several locations are believed to have reduced or partially replaced native bivalves, including *Macoma nasuta* (Bent-Nose Macoma) in San Francisco Bay (Cohen et al., 1995), *Macoma balthica* in the Baltic Sea (Obolewski and Piesik, 2005), *Lentidium mediterraneum* in the Black Sea (Skolka and Preda, 2010), and *Cerastoderma edule* (Edible Cockle) in the Skagerrak, Sweden (Möller, 1986). During periods of exceptional abundance, *M. arenaria* may have effects throughout the food web, affecting phytoplankton abundance, and in turn, zooplankton, mysids, and fish recruitment.

Where it is abundant, it is an important suspension-feeder, grazing phytoplankton, and an important food item for fishes, invertebrates, and birds (Nichols and Thompson, 1985b, Zaiko et al., 2011). It is also a potential competitor with native bivalves (Möller, 1986, Conde et al., 2011).

Estimated feeding rates of *M. arenaria* in the southwestern Baltic Sea, off Germany, indicate that this clam can filter the entire water column once or several times a day, depending on water depth (Forster and Zettler, 2004). Large biomasses in San Francisco Bay (Nichols and Thompson, 1985b, a), the Skagerrak (Möller, 1986), the Baltic (Bubinas and Vaitonis, 2003, Forster and Zettler, 2004, Obolewski and Piesik, 2005, Zaiko et al., 2011), and Black Sea (Gomoiu et al., 2002) imply significant feeding rates.

Mya arenaria, when abundant, has been an important prey organism for clam worms (Nereidae), predatory snails, shrimps, crabs, fishes, ducks, and shorebirds in invaded regions (Carlton, 1979, Sadykhova, 1979, Ozturk, 2002, Bubinas and Vaitonis, 2003, Cloern et al., 2007, Skolka and Preda, 2010). Because it tolerates low salinities and wide tidal ranges better than many native clams, it has the potential to increase the food supply for predators in estuaries." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60).

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mya arenaria* would have high impact in many areas on communities in the RA area. Clear major effects on various populations and the community in general are well documented.

Background Information: "During periods of exceptional abundance, *M. arenaria* may affect phytoplankton abundance, and in turn, zooplankton, mysids, and fish recruitment. High abundances of *M. arenaria* during 'boom' periods, or its empty shells during 'busts,' can affect the abundance of predators with implications for other benthic organisms. For example, high abundances of *M. arenaria* shells supported elevated abundances of juvenile Dungeness Crabs (*Metacarcinus magister*) in Grays Harbor, WA which could lead to increased predation on other benthic organisms (Palacios et al., 2000).

From NEMESIS: "When abundant, *M. arenaria* is a significant herbivore in estuaries, because of its large size and powerful filtration, and its ability to survive in low salinities and wide tidal ranges where large native bivalves are often rare.

Introduced populations of *M. arenaria* in several locations are believed to have reduced or partially replaced native bivalves, including *Macoma nasuta* (Bent-Nose Macoma) in San Francisco Bay (Cohen et al., 1995), *Macoma balthica* in the Baltic Sea (Obolewski and Piesik, 2005), *Lentidium mediterraneum* in the Black Sea (Skolka and Preda, 2010), and *Cerastoderma edule* (Edible Cockle) in the Skagerrak, Sweden (Möller, 1986). Where it is abundant, it is an important suspension-feeder, grazing phytoplankton, and an important food item for fishes, invertebrates, and birds (Nichols and Thompson, 1985b, Zaiko et al., 2011). It is also a potential competitor with native bivalves (Möller, 1986, Conde et al., 2011).

Estimated feeding rates of *M. arenaria* in the southwestern Baltic Sea, off Germany, indicate that this clam can filter the entire water column once or several times a day, depending on water depth (Forster and Zettler, 2004). Large biomasses in San Francisco Bay (Nichols and Thompson, 1985b, a), the Skagerrak (Möller, 1986), the Baltic (Bubinas and Vaitonis, 2003, Forster and Zettler, 2004, Obolewski and Piesik, 2005, Zaiko et al., 2011), and Black Sea (Gomoiu et al., 2002) imply significant feeding rates.

Mya arenaria, when abundant, has been an important prey organism for clam worms (Nereidae), predatory snails, shrimps, crabs, fishes, ducks, and shorebirds in invaded regions (Carlton, 1979, Sadykhova, 1979, Ozturk, 2002, Bubinas and Vaitonis, 2003, Cloern et al., 2007, Skolka and Preda, 2010). Because it tolerates low salinities and wide tidal ranges better than many native clams, it has the potential to increase the food supply for predators in estuaries." (https ://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60).

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mya arenaria* would have high impact in many areas on habitat in the RA area. There are well documented major effects on the habitat in general.

Background Information: From NOBANIS: "*Mya arenaria* seems to often die *in situ*, forming so-called death assemblages (Strasser, 1998), which can persist for maybe 100 years or more and form habitats for other species (Palacios et al., 2000). *Mya arenaria* has a high capacity for filtration. Volumes of about one to ten liters per hour have been measured for clams of 6-7cm shell length (Jørgensen and Riisgård, 1988), and population filtration rates of more than 8 m^3 m^{-2} day^{-1} have been calculated in the southern Baltic Sea (Forster and Zettler, 2004)" (https://www.nobanis.org/globalassets/speciesinfo/m/mya-arenaria/mya-arenaria.pdf).

From NEMESIS: "Softshell clams are suspension feeders and can burrow up to 20 cm (in large specimens), with their siphon protruding above the surface. *Mya arenaria*, as a powerful burrower and filterer, has the potential to alter habitats and sediment characteristics through bioturbation and deposition of pseudofeces and also through suspension feeding, increasing water clarity, and light penetration (Obolewski and Piesik, 2005, de Moura Queirós et al., 2011, Zaiko et al., 2011). Introduced populations of *M. arenaria* have often gone through boom-and bust phases, leaving 'death assemblages' of empty shells, providing habitat for many other benthic organisms (Strasser, 1998, Palacios et al., 2000)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60).

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mya arenaria* would have high impact in many areas on ecosystem function in the RA area. There are well documented major effects on ecosystem structure in general.

Background Information: In Ringkøbing Fjord the invasion of *M. arenaria* caused a decrease in chlorophyll a to 17% of previous concentrations (Petersen et al., 2008). (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60). *Mya arenaria* can cause regime shifts: changes in dominating pathways of organic matter production from pelagic turnover to benthic pelagic coupling (Petersen et al., 2008).

From NEMESIS: "During periods of exceptional abundance, *Mya arenaria* may have effects throughout the food web, affecting phytoplankton abundance, and in turn, zooplankton, mysids, and fish recruitment. This may have happened in 1976-1977 in Suisun Bay, California (Nichols and Thompson, 1985a, Cohen et al., 1995). High abundances of *Mya arenaria* during 'boom' periods, or its empty shells during 'busts,' can affect the abundance of predators with implications for other benthic organisms. For example, high abundances of *M. arenaria* shells supported elevated abundances of juvenile Dungeness Crabs (*Metacarcinus magister*) in Grays Harbor, WA which could lead to increased predation on other benthic organisms (Palacios et al., 2000)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60)

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Mya arenaria* would have low or no impact associated with diseases, parasites or travelers in the RA area. The only likely stage of transport are veligers in ballast – unlikely these would be infected by *Perkinsus* sp. or other diseases and parasites. However, this is uncertain based on the literature.

Background Information: Mya arenaria is a non-host filter feeder filtering cercariae of the *Himasthla elongate* trematode, but not becoming infected (Thieltges et al., 2008). The species' 1st intermediate host (*Littorina littorea*) is not present in HBC and thus there is no risk from this parasite as it cannot complete its life cycle.

From Dungan et al. (2002): "It can be affected by the protozoan *Perkinsus* sp. (Mild pathology and prevalent defensive parasite encapsulation observed in some infections are interpreted to suggest only that they may compromise growth and reproduction of infected clams (McLaughlin and Faisal, 1998). However, high parasite densities and systemic distributions shown for other clams indicate an acute, probably lethal, disease condition. Sampled populations of *M. arenaria* in Chesapeake Bay were infected by *Perkinsus* sp. Protozoans at prevalence ranging from 30 to 100% of sampled clams."

From Reece et al. 2008: "*Perkinsus marinus* and *Perkinsus chesapeaki* have been found in other oysters and clams (*Crassostrea virginica*, *Macoma balthica*, *Tagelus plebeius*, *Mulina lateralis*, *Rangia cuneata*, *Cyrtopleura costata*)." (*Macoma balthica* is present in the risk assessment region).

From Strasser (1998): "Several diseases and parasites are known to occur in *Mya* on the Atlantic west coast (Gibbons and Blogoslawski, 1989, Hidu and Newell, 1989; and references therein). *Mya* may be heavily affected by two types of cancer called disseminated neoplasia and germinomas (Landsberg, 1996). Clams categorized into high-severity neoplasia suffered mortalities of up to 78% in *Mya* in New England (Brousseau and Baglivo, 1991). On the Atlantic east coast no parasites have been found in *Mya* from the German Wadden Sea (Lauckner, personal communication)."

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Mya arenaria* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native clams of the same genus in the RA area, but no evidence of hybridization is found in the literature.

Background Information: Mya pseudoarenaria and *Mya truncata* are native species in the assessment area (Stewart and Lockhart, 2005, Roy and Gagnon, 2016).

From Zhang et al. (2018): "While there were significant differences in sperm morphology between *M. arenaria* and *M. japonica*, we are uncertain if these differences (or ecological/behavioural differences) would prohibit hybridization." [Information suggests possibility of *M. arenaria* being known to hybridize with other species in the same genus, but with low certainty; other species of the same genus may be present in the region but it does not mean that they can hybridize].

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Mya arenaria* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. Effects are possible but there is little supporting evidence.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and at-risk fish and most mammal species are unlikely to be affected. Eskimo Curlew (*Numenus borealis*), which is an endangered species (Stewart and Lockhart, 2005) (but has not been observed since the 1980s and its range did not include HBC) can act as a predator for *M. arenaria* so it could even have a positive effect on it. Depending on impacts to *M. truncata*, which is one of walrus' main foods, it may have potential impact on their populations.

Q16- Impact on fisheries (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mya arenaria* would have high impact in many areas on fished species in the RA area. It is likely to impact the abundance of the locally harvested benthic species (especially other *Mya* spp.) and possibly plankton feeding species given known impacts elsewhere.

Background Information: From NEMESIS: “Introduced populations of *M. arenaria* in several locations are believed to have reduced or partially replaced native bivalves, including *Macoma nasuta* (Bent-Nose Macoma) in San Francisco Bay (Cohen et al., 1995), *Macoma balthica* in the Baltic Sea (Obolewski and Piesik, 2005), *Lentidium mediterraneum* in the Black Sea (Skolka and Preda, 2010), and *Cerastoderma edule* (Edible Cockle) in the Skagerrak, Sweden (Möller, 1986).

During periods of exceptional abundance, *Mya arenaria* may have effects throughout the food web, affecting phytoplankton abundance, and in turn, zooplankton, mysids, and fish recruitment.” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60)

Mya truncata is fished in the RA (Igoolik Hunters and Trappers, pers comm) and may be displaced by *M. arenaria* in lower intertidal areas. That said, *M. arenaria* is also fished throughout its range and may also be fished if it becomes established, potentially with major impacts on fisheries/harvesting. Likewise, other shallow water benthic species could be impacted by competition/food web effects and/or habitat/ecosystem effects (Kotta et al., 2001, Kotta et al., 2003). Mussels (*Mytilus* sp.), sea urchin (*Strongylocentrotus droebachiensis*), brown sea cucumber (*Cucumaria japonica*), Iceland scallops (*Chlamys islandica*) are also harvested by Inuit in the RA and could be impacted (Stewart and Lockhart, 2005).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mya arenaria* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: From NEMESIS: “They were introduced to San Francisco Bay between 1869 and 1874 with oysters from the east coast and have since expanded their distribution northward to ca. Kotzebue Sound, AK (70°N) (Fofonoff et al., 2003). *M. arenaria*’s current native range is from subarctic Labrador, Canada to Cape Hatteras, North Carolina and sporadically to South Carolina (Abbott, 1974, Gosner, 1978), and in the Northwest Pacific, from the Yellow Sea, China to the Bering Sea (formerly as *M. japonica*) (Zenkevich, 1963; Golikov, 1976). However, based on the fossil record, this species originated in the North Pacific Ocean, possibly around Japan, during the Miocene period and soon colonized the Atlantic, reaching the European coast in the late Pliocene, but then dying out during most of its range in the Pleistocene. In Europe, the West Coast, and Alaska, it is absent for prehistoric human shell middens, disregarding some probable misidentifications (Carlton, 1979). The surviving populations were on the East Coast of North America, and the East Coast of Asia (Vermeij, 1989, Strasser, 1998). *M. arenaria* appears to be extinct in the Arctic Ocean, though determining its present distribution is complicated by occurrence of subfossil shells and other species of *Mya* and related genera (Bernard, 1979; James T. Carlton, pers. Comm.). Humans have re-introduced *M. arenaria* to much of its former range, and beyond. Vikings may have transported this clam to Scandinavia as early as the 13th century, and later shipping and food introductions may have moved it to most of the European coast, from the Barents Sea to the Iberian Peninsula (Petersen et al., 1992, Strasser, 1998). It is also established in a few estuaries along the Mediterranean Sea (Zenetos et al., 2003) and in the Black Sea (Gomoiu et al., 2002). Softshell Clams were apparently introduced to the West Coast with plantings of Eastern Oysters (*Crassostrea virginica*) by 1874, and were soon deliberately transplanted as food as far north as Alaska (Carlton, 1979, Powers et al., 2006). Recent genetic studies support the recent (post-Pleistocene) introduction of *Mya arenaria* to Europe and the West Coast of

North America (Cross et al., 2016, Lasota et al., 2016)”.
(https ://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-60)

ZOOBENTHOS – TUNICATES

Botrylloides violaceus Oka, 1927

Phylum: Chordata
Class: Ascidiacea
Order: Stolidobranchia
Family: Styelidae

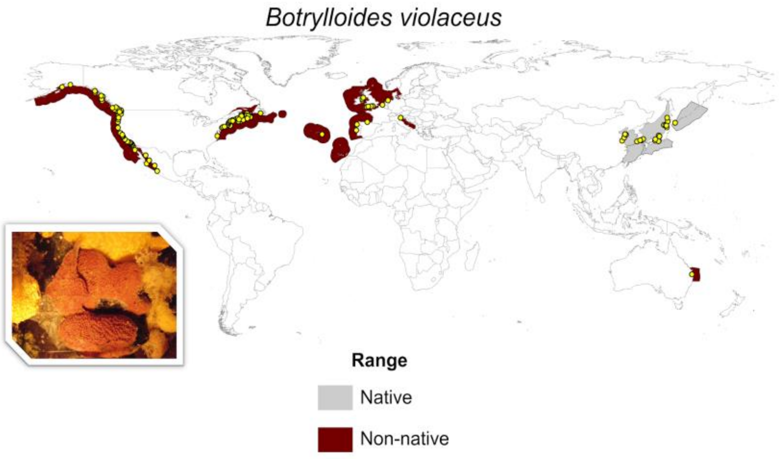

Figure 15: Ecoregions where Botrylloides violaceus is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution, and occurrence points (in yellow) have not been collected in an exhaustive manner. Botrylloides violaceus occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://api.gbif.org/v1/occurrence/download/request/0005436-141123120432318.zip, 15 December 2014), NEMESIS (http://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-100), Kott (2003), Vaz-Pinto et al. (2014). Picture of B. violaceus modified from https://invasions.si.edu/nemesis/species_summary/-100.

CMIST scores for *B. violaceus:*

Mean adjusted Likelihood of Invasion: 2.00

Mean adjusted Impact of Invasion: 1.86

Mean adjusted Overall CMIST score: 3.72

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Botrylloides violaceus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports of the species were found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). It is present north to the Maritimes and Gulf of St Lawrence along the Atlantic coast and to Alaska on the Pacific coast (Carver et al., 2006a) (DFO 2012).

Q2- Rate of introduction **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Botrylloides violaceus* could arrive frequently in low numbers or infrequently in high numbers in the RA area. The species is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Hull fouling is the most likely transport vector for *B. violaceus* into Hudson Bay, although transport of colony fragments in ballast water or on fouled fishing gear is possible. Propagule pressure is unknown but *B. violaceus* is present at ports directly connected to Churchill, Deception Bay, and Iqaluit by shipping (Chan et al., 2012). Successful transport by other vectors is unlikely since there is no aquaculture or commercial marine fishing at present, the pelagic larval cycle is short for transport by ballast water (Carlton and Geller, 1993), and transport from Atlantic Canada by natural dispersal (e.g., on rafts of eelgrass or floating debris) or by recreational boating is unlikely due to currents and distance.

Botrylloides violaceus is a common fouler on pleasure craft and fishing vessels (Simard et al., 2017), and common in ecoregions linked to the HB LME by shipping and pleasure craft.

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Botrylloides violaceus*. There is reliable information in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support habitat classification.

Background Information: This epibenthic species occurs in sheltered areas often on artificial substrates such as floating docks, wharf pilings, and boat hulls and also on natural substrates such as rocky reefs, bivalve colonies, seaweeds, and eelgrass (Simkanin et al., 2012, Wong and Vercaemer, 2012, Carman et al., 2016). Dock floats are especially favored habitats, probably because their motion provides rapid water exchange, and a fresh supply of food-laden water (Glasby, 2001). Colonies are generally restricted to zones <50 m deep (Therriault and Herborg, 2008b). They are susceptible to desiccation and rarely observed in intertidal areas unless damp and shaded (Rinkevich et al., 1993).

Q4- Survival (suitable environment) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Botrylloides violaceus*. Temperature tolerance of -0.6°C, species can survive in other locations (e.g., Gulf of St. Lawrence) under ice cover at sub-zero temperatures. These characteristics are at the extreme northern extent of its distribution. Winters would be shorter and with thinner ice cover, so suitable environment would be expected through some parts of the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Botrylloides violaceus is a cold temperate to warm temperate, polyhaline to euryhaline species (Fofonoff et al., 2003). Laboratory-raised juvenile colonies were studied using a two-factorial experimental design with five levels of temperature (5, 10, 15, 20, 25 °C) and five levels of salinity (14, 20, 26, 32, 38‰) (Epelbaum et al., 2009a). *Botrylloides violaceus* tolerated a broad range of temperatures (5-25 °C) and salinities (20-38‰), grew at 15-25 °C and 26-38‰, and attained its largest colony sizes at 20-25 °C and 26-38‰. The low temperature limit for juvenile colonies of *B. violaceus* is between 0 and 5 °C (Epelbaum et al., 2009a). The species is present north to the Maritimes and Gulf of St Lawrence along the

Atlantic coast and to Alaska on the Pacific coast (Carver et al., 2006a, DFO 2012) where waters reach sub-zero temperatures. At Belleoram, Newfoundland near the present northern distribution limit of established *B. violaceus* in Atlantic Canada the monthly average water temperatures ranged from 1.0 ºC in February to 16.4 ºC in August (McKenzie et al., 2016).

Zerebecki and Sorte (2011) suggested a low temperature tolerance of -0.6 °C, based on the species' presence at Anchorage, AK and temperature data from the World Ocean Atlas which may not equal conditions experienced by *B. violaceus* in situ. Niche modelling results showed that projected environmental conditions under global warming will include a small region of the assessment area as suitable habitat (Goldsmit et al., 2018).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Botrylloides violaceus* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with a maximum sea surface temperature >=5°C, the lowest limit from which experimentally tested colonies could reverse hibernation and develop functional zooids. Some evidence suggests colonies may survive and recruit at lower temperatures, down to 0 °C.

Background Information: Botrylloides violaceus is a viviparous species that can reproduce sexually and asexually (Carver et al., 2006a; and references therein). In PEI, assuming a 1-month maturation period for *B. violaceus* larvae, egg fertilization likely occurred in mid-June when water temperatures reached 15°C. Larval recruitment was ongoing until mid-October or approximately 1-month after the cessation of egg production. Larvae swim for a brief period (4 to 10 h) before attaching to a suitable substrate to start their own colony or fuse with an existing colony. The mean number of zooids exceeded 100 per colony within 2 weeks at 20 to 25°C and 4 weeks at 14 to 20°C. Farther south, in New England, growth and reproduction of *B. violaceous* colonies slows down but continues through the winter (E. Westerman, Univ. of Arkansas, personal communication): "In at least one year between 2007-2012, Bob Whitlatch's lab at Stores (UConn) found *B. violaceous* recruits in February, which means the parent colonies were brooding larvae in January, at temperatures between 0-10°C."

The low temperature limit for juvenile colonies of *B. violaceus* is between 0 and 5°C (Epelbaum et al., 2009a). Those held in the laboratory at 0°C went through irreversible degeneration and died within the first two weeks of the experiment (Epelbaum et al., 2009a). In 5°C treatments (at 20, 26, 32, and 38‰) all colonies underwent morphological changes similar to hibernation; zooids and buds regressed while ampullae proliferated and covered the regressing structures. Hibernating colonies remained in that state until the end of the experiment. To test whether hibernation was reversible, 24 colonies at 5°C (six per salinity treatment) were gradually acclimated to 10°C (1°C every 2 h). All 24 colonies recovered from hibernation and developed functional zooids within four days.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Botrylloides violaceus*. Several studies show species-specific effects, but the findings are inconsistent.

Background Information: Predation may limit or slow the spread of *B. violaceus* to natural habitats (Simkanin et al., 2013), especially in open coastal areas (Osman and Whitlatch, 2004). Consumers include fish, snails, chitons, crabs, urchins, and starfish; competitors include colonial tunicates and bryozoans (Fofonoff et al., 2003, Nydam and Stachowicz, 2007). Fish predate recruits during the first week or so after settlement but seldom predate the adults (Carver et al., 2006a; and references therein). Observations of partially eaten recruits suggested they are unpalatable or chemically defended like some other ascidians. Chitons also predate newly settled recruits (Nydam and Stachowicz, 2007). The patchy distribution of many ascidians may reflect spatial variability in the predator abundance (Carver et al., 2006a). Further information is needed on possible control agents.

From Dorning (2017): “Predators: No natural predators observed, possibly due to chemical unpalatability. Only vulnerable to potential predation a short period after settlement (days to a week) (Pisut and Pawlik, 2002, Tarjuelo et al., 2002). Experimental exclusion of potential chiton, gastropod and flatworm predators does not affect *B. violaceus* recruitment or abundance (Grey, 2010). Potential sea star, crab, nudibranch, and urchin predators prefer natural prey to *B. violaceus* (Epelbaum et al., 2009b). Ascidian predators include fish, crab, polychaetes, sea stars; especially prosobranch molluscs, opisthobranchs, nudibranchs, turbellarian flatworms and the grey seal (Scotland). Used by man for food (Japan, Mediterranean, Chile) and bait (Australia, South Africa). Destroyed as a pest in oyster beds and commercial fishing grounds (Millar, 1971).”

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Botrylloides violaceus* has the capacity for a moderate range of natural dispersal in the RA area. There is some evidence to suggest a reasonable degree of natural dispersal for asexual buds attached to objects, however the short larval duration limits dispersal at this life stage.

Background Information: Natural dispersal can occur for both asexual buds and tadpole larvae. The short duration of the free-swimming larval stage (<24 h at 16-23 °C) (Takeuchi, 1980, Carver et al., 2006a) limits larval dispersal by swimming. Other natural vectors for dispersal include rafting on eelgrass, algae or other forms of floating debris (Lambert and Lambert, 2003); or as epibionts on large crustaceans that move locally or are exported globally (Pederson et al., 2005, Bernier et al., 2009).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence for wide range dispersal through biofouling, which could occur on vessels visiting the RA area as well as on fishing and recreational vessels that move within the region.

Background Information: Anthropogenic dispersal can occur for both asexual buds and tadpole larvae. Molecular data indicate that colonial ascidians disperse predominantly through sexually produced propagules but practices that fragment colonies, such as high pressure washing, potentially exacerbate infestations and spread via asexual propagules (Bock et al., 2011). Hull fouling by sessile stages of *B. violaceous*, particularly on slower moving vessels such as barges, is the most likely vector for introducing the species into the region and spreading it within the region (Lambert and Lambert, 2003), although it could also be spread by recreational boats (Clarke Murray et al., 2011). The short duration of the free-swimming larval stage (<24 h at 16-23 °C) (Takeuchi, 1980) makes transport of free-swimming larvae in ballast water over long distances unlikely (Lambert and Lambert, 2003). However, ballast often contains floating bark chips, seaweed, and other particulates that could have sessile *B. violaceous* attached. Commercial fishing and aquaculture-related activities have also been responsible for the introduction of colonial tunicates (Lambert, 2005) but these activities are limited in Hudson Bay Complex. The importance of potential dispersal vectors in the RA area other than shipping (e.g., smaller pleasure and fishing vessels) is not well known.

Q9- Impact on population (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. Reliable information exists on impacts to populations in invaded locations. Effects are largely limited to artificial structures so strong effects would only be likely in a few areas of the RA area.

Background Information: Botrylloides violaceus is a common fouling organism throughout much of its introduced range (Fofonoff et al., 2003; and references therein; Pederson et al., 2005, Carver et al., 2006a). It frequently displaces other fouling organisms, including native and introduced tunicates, bryozoans, barnacles, and mussels through competition for space and food.

Surface fouling by *B. violaceus* could reduce productivity and coverage of native eelgrass (*Zostera marina*) beds (Wong and Vercaemer, 2012). The violet morph of *B. violaceus* has a greater negative effect than the lighter-colored tunicates, which transmit more light through their bodies. Eelgrass beds in James Bay offer key habitat for many marine species and browse for large populations of migratory waterfowl (Stewart and Lockhart, 2005).

Zajac et al. (1989) noted that *Botrylloides* spp. Had a negative impact on the survival and growth of oyster spat probably as a result of localized food depletion rather than overgrowth. Arakawa (1990) noted that tunicates grew more rapidly than oyster spat and effectively interfered with their survival. Anecdotal reports from the Bras D'Or Lakes (N.S.) also suggested that *B. schlosseri* can overgrow and smother young oyster spat on shell collectors (Stuart, pers. comm.). In the case of juveniles and adults, the potential impacts of tunicate smothering behaviour vary among studies. For example, Dalby Jr and Young (1993) observed no consistent pattern in terms of the impact of colonial ascidians on oysters; overgrowth did not always lead to mortality and in some cases growth and survival were enhanced. In comparative trials in P.E.I. where *B. violaceus* was actively cleaned from the surface of cultured mussels, MacNair (pers. comm.) reported no significant positive impact on growth, meat yield or survival relative to heavily fouled mussels. Sebens et al. (1997) noted in extensive diving surveys that the combined presence of sea urchins and *Botrylloides* severely impacted the indigenous assemblage, and the botrylloids seemed immune to urchin grazing. It was argued that, even in the absence of the urchins, the capacity of *Botrylloides* to overgrow and outcompete indigenous species may have compromised the former natural habitat.

Less well-documented chemical interactions include the observation that *B. schlosseri* metabolites may have toxic effects on barnacle nauplii, copepods and mussel gills (Braiko et al., 1988). Also, Teo and Ryland (1995) suggested that extracts of *B. schlosseri* displayed moderate levels of toxicity against invertebrate larvae. In contrast, Keough (1998) found that the presence of *B. schlosseri* had a positive impact on the settlement of other fouling organisms, specifically arborescent bryozoans (*Bugula* spp.).

Q10- Impact on communities (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* would have high impact in few areas or moderate impact in many areas on communities in the RA area. Reliable information exists on impacts to communities in invaded locations. Effects are largely limited to artificial structures, so strong effects would only be likely in a few areas of the RA area.

Background Information: Botrylloides violaceus is a common fouling organism throughout much of its introduced range (Fofonoff et al., 2003; and references therein; Pederson et al., 2005, Carver et al., 2006a). It frequently displaces other fouling organisms, including native and introduced tunicates, bryozoans, barnacles, and mussels through competition for space and food. By altering plant, invertebrate and possibly fish communities it could impact species dominance and diversity, and community dynamics (see also Dijkstra and Harris, 2007, Dijkstra and Harris, 2009).

From Carver et al. (2006a): “Colonial tunicates compete for space by overgrowing and smothering existing species; in some cases the net impact may be a reduction in community species diversity. In a rapid assessment survey of native and non-native marine species of floating dock communities in New England, Pederson et al. (2005) found that *B. violaceus* and *B. schlosseri*, along with two other introduced compound ascidians, were very effective at overgrowing algae and other fouling organisms. These tunicate species represented 25% of the introduced species, whereas tunicates in general accounted for only 4% of all species.

Zajac et al. (1989) noted that *Botrylloides* sp. Had a negative impact on the survival and growth of oyster spat probably as a result of localized food depletion rather than overgrowth. Arakawa (1990) noted that tunicates grew more rapidly than oyster spat and effectively interfered with their survival. Anecdotal reports from the Bras D’Or Lakes (N.S.) also suggested that *B. schlosseri* can overgrow and smother young oyster spat on shell collectors (Stuart, pers. Comm.). In the case of juveniles and adults, the potential impacts of tunicate smothering behaviour vary among studies. For example, Dalby Jr and Young (1993) observed no consistent pattern in terms of the impact of colonial ascidians on oysters; overgrowth did not always lead to mortality and in some cases growth and survival were enhanced. In comparative trials in P.E.I. where *B. violaceus* was actively cleaned from the surface of cultured mussels, MacNair (pers. Comm.) reported no significant positive impact on growth, meat yield or survival relative to heavily fouled mussels.

Sebens et al. (1997) noted in extensive diving surveys that the combined presence of sea urchins and *Botrylloides* severely impacted the indigenous assemblage, and the botrylloids seemed immune to urchin grazing. It was argued that, even in the absence of the urchins, the capacity of *Botrylloides* to overgrow and outcompete indigenous species may have compromised the former natural habitat.”

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species impacts habitat structure via overgrowth/competition in invaded areas. Effects are largely limited to artificial structures, so strong impacts would only be likely in a few areas of the RA area.

Background Information: Introduction of colonial tunicates can reduce habitat availability for other species by covering natural and artificial surfaces (Carver et al., 2006a). Based on the results of its introduction elsewhere (Carver et al., 2006a), *B. violaceus* could negatively impact habitat structure by overgrowing and smothering existing species. For example, when *B. violaceus* replaces the blue mussel (*Mytilus edulis*) as the dominant species in a fouling community a major functional habitat change occurs (Dijkstra and Harris, 2009). The year-round structure mussels provide for other organisms to settle upon (secondary settlement) is lost, and seasonal die off of *B. violaceus* creates large areas of bare substrate for organisms to colonize. Habitat degradation can occur if there is increased fecal deposition or widespread die-off that lead to an accumulation of decaying biomass on the bottom (Carver et al., 2006a). Under extreme conditions, water quality could be impacted by ammonia excreted by the tunicate colonies. In substantial densities colonial tunicates may increase water clarity by particle removal. This can negatively impact resident filter feeders such as zooplankton and bivalve populations by reducing the amount of food available, but improve light transmission through the water column, enhancing benthic plant production.

Q12- Impact on ecosystem function (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. The species may impact ecosystem function via overgrowth/competition in invaded areas and perhaps altering water turbidity (and plankton loads). These effects are largely limited to artificial structures, so strong effects would only be likely in a few areas of the RA area.

Background Information: Based on the results of its introduction elsewhere (Carver et al., 2006a), *B. violaceus* could negatively impact the food web and trophic structure of invaded ecosystems by inducing changes in plant, invertebrate and possibly fish communities (Therriault and Herborg, 2008b; and references therein; Wong and Vercaemer, 2012). Under extreme conditions, water quality could be impacted by the amount of ammonia excreted by the tunicate colonies (Carver et al., 2006a).

From Carver et al. (2006a): "The introduction of a colonial tunicate species may have a negative impact on water quality depending on the size of the population and the hydrodynamics of the system. Under turbid conditions the increased rate of particle removal may improve water clarity thereby enhancing benthic macroalgal or eelgrass production. If food resources are limited, however, this increased demand may negatively impact resident filter feeders such as zooplankton and bivalve populations. A substantial increase in tunicate biomass may translate into greater biodeposition of fecal material that may in turn increase the risk of benthic habitat degradation. This problem may be exacerbated if colonial tunicates undergo a widespread mortality event and the decaying biomass accumulates on the bottom. Under extreme conditions, water quality could be impacted by the amount of ammonia excreted by the tunicate colonies.

Colonial tunicates compete for space by overgrowing and smothering existing species; in some cases the net impact may be a reduction in community species diversity. In a rapid assessment survey of native and non-native marine species of floating dock communities in New England, Pederson et al. (2005) found that *B. violaceus* and *B. schlosseri*, along with two other introduced compound ascidians, were very effective at overgrowing algae and other fouling organisms. These tunicate species represented 25% of the introduced species, whereas tunicates in general accounted for only 4% of all species."

Zajac et al. (1989) noted that *Botrylloides* sp. had a negative impact on the survival and growth of oyster spat probably as a result of localized food depletion rather than overgrowth. Arakawa (1990) noted that tunicates grew more rapidly than oyster spat and effectively interfered with their survival. Anecdotal reports from the Bras D'Or Lakes (N.S.) also suggested that *B. schlosseri* can overgrow and smother young oyster spat on shell collectors (Stuart, pers. Comm.). In the case of juveniles and adults, the potential impacts of tunicate smothering behaviour vary among studies. For example, Dalby Jr and Young (1993) observed no consistent pattern in terms of the impact of colonial ascidians on oysters; overgrowth did not always lead to mortality and in some cases growth and survival were enhanced. In comparative trials in P.E.I. where *B. violaceus* was actively cleaned from the surface of cultured mussels, MacNair (pers. Comm.) reported no significant positive impact on growth, meat yield or survival relative to heavily fouled mussels.

Less well-documented chemical interactions include the observation that *B. schlosseri* metabolites may have toxic effects on barnacle nauplii, copepods and mussel gills (Braiko et al., 1988). Also, Teo and Ryland (1995) suggested that extracts of *B. schlosseri* displayed moderate levels of toxicity against invertebrate larvae. In contrast, Keough (1998) found that the presence of *B. schlosseri* had a positive impact on the settlement of other fouling organisms, specifically arborescent bryozoans (*Bugula* spp.)."

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Botrylloides violaceus* would have low or no impact associated with diseases, parasites or travelers in the RA area.

Rates of parasitism in ascidians are generally high and there is evidence of a lethal disease that can be carried by the species, but little evidence of impacts.

Few studies are available on diseases and parasites that affect *B. violaceus* (Carver et al., 2006a, Therriault and Herborg, 2008b). "Cup Cell Disease", an infectious disease associated with haplosporidian protists, is lethal to *B. violaceus* (Moiseeva et al., 2004). Its potential impacts in the event of introduction are unknown.

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Botrylloides violaceus* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area.

Background Information: Botrylloides aureum has been reported from Hudson Strait and Foxe Basin (Stewart and Lockhart, 2005, Goldsmit et al., 2014) (https://www.gbif.org, https://obis.org/); whether it might hybridize with *B. violaceus* is unknown.

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* would have low or no impact on at-risk or depleted species in the RA area.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay; at-risk fish, bird, and mammal species are unlikely to be affected. Areas of overlap would be few as introduced *B. violaceus* would likely be associated with manmade structures, which are few.

Q16- Impact on fisheries (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Botrylloides violaceus* would have low or no impact on fished species in the RA area. Areas of overlap would be few as introduced *B. violaceus* would likely be associated with manmade structures, which are few. Potential effects related to *B. violaceus* fouling of eelgrass beds are uncertain.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Fouling by colonial tunicates entirely covered some cultured blue mussels in PEI but valve openings were maintained, allowing continued siphoning by the mussels. The fouling had little effect on the growth and production of adult mussels (Arens et al., 2011) but might affect larval settling and smother spat. Commercially harvested anadromous Arctic charr (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals. Surface fouling by *B. violaceus* could reduce productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). In James Bay, these plants offer habitat for many marine species and browse for large populations of migratory waterfowl that are hunted locally for subsistence and sport (Stewart and Lockhart, 2005).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botrylloides violaceus* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Botrylloides violaceus is native to the Northwest Pacific from northern Japan to southern Korea and northern China (Fofonoff et al., 2003, Simkanin et al., 2016; and references therein from Fofonoff). It has been widely introduced to the Northeast Pacific, the Northwest Atlantic, and parts of the Northeast Atlantic. On the West Coast of North America, it was first discovered in Santa Barbara, California in 1966 and has since spread along the coast from Mexico to Alaska; to date it has failed to establish in Alaska. On the East Coast, it was found in New England in the late 1970s and can now be found from Chesapeake Bay to Newfoundland (see also Carver et al., 2006a, McKenzie et al., 2016). Genetic studies found that non-native populations in Washington and BC were established by multiple primary introduction events from the species native range in Asia; whereas those in Eastern Canada appear to have spread by contiguous stepping-stone movements through secondary introduction vectors (Bock et al., 2011). Due to its abundance, wide distribution, and frequent dominance, this ascidian is likely to have substantial impacts on shipping, aquaculture, and fisheries (Fofonoff et al., 2003; and references therein).

Botryllus schlosseri **(Pallas, 1766)**

Phylum: Chordata

Class: Ascidiacea

Order: Stolidobranchia

Family: Styelidae

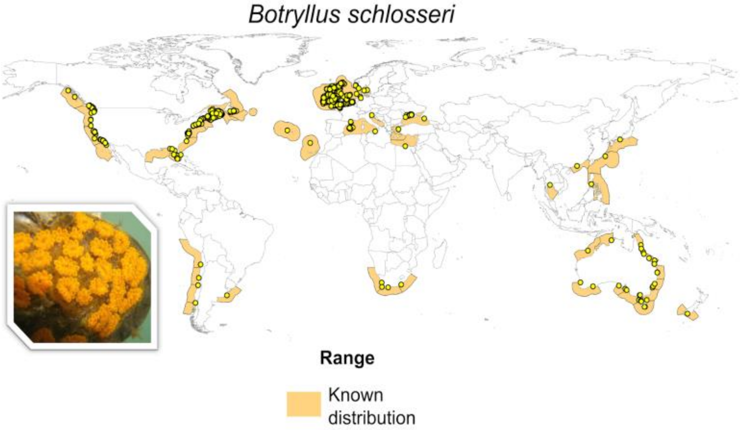

Figure 16: Ecoregions where *Botryllus schlosseri* *is distributed (orange). As it is a cryptic species, its origin and species status remain uncertain. These regions are only representative of the main known distribution and occurrence points (in yellow) were not collected in an exhaustive manner.* *Botryllus schlosseri* *occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.0xnrga, 27 June 2017), Ben-Shlomo et al. (2010), Canning-Clode et al. (2013), Turon et al. (2016), Mead et al. (2011), and Fofonoff et al. (2003). Picture of* *B. schlosseri* *modified from* *https://invasions.si.edu/nemesis/species_summary/159373*.

CMIST scores for *B. schlosseri:*

Mean adjusted Likelihood of Invasion: 2.02

Mean adjusted Impact of Invasion: 1.76

Mean adjusted Overall CMIST score: 3.55

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Botryllus schlosseri* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports of *B. schlosseri* were found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). The species is present north to the Maritimes and Gulf of St Lawrence along the Atlantic coast and on the Pacific coast of Canada and the US (Carver et al., 2006a, Deibel et al., 2014).

Q2- Rate of introduction **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Botryllus schlosseri* could arrive frequently in low numbers or infrequently in high numbers in the RA area. The species is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Hull fouling is the most likely transport vector for *B. schlosseri* into Hudson Bay, although transport of colony fragments in ballast water or on fouled fishing gear is possible. Successful transport by other vectors is unlikely since there is no aquaculture at present. The pelagic larval cycle is short for transport by ballast water (Carlton and Geller, 1993), and transport from Atlantic Canada by natural dispersal (e.g., on rafts of eelgrass or floating debris) or by recreational boating is unlikely due to currents and distance.

Therriault and Herborg (2008b) ranked the arrival of *B. schlosseri* on the Atlantic coast of Canada as very high with very low uncertainty, based on vectors including ship fouling, aquaculture/fisheries gear, and dispersal. Of these vectors only hull fouling is likely to transport the species into Hudson Bay.

Propagule pressure is unknown but *B. schlosseri* is present at ports directly connected to Churchill and Deception Bay by shipping (Chan et al., 2012).

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Botryllus schlosseri.* There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the habitat classification.

Background Information: *Botryllus schlosseri* survives from the low intertidal/subtidal zone to 200 m depth on natural (rocky reefs, bivalve colonies, seaweeds, eelgrass) and artificial substrates (docks, pilings, aquaculture gear, vessels) (Fofonoff et al., 2003, Carver et al., 2006a, Deibel et al., 2014). It has a particular affinity for artificial structures such as harbour and aquaculture infrastructure although such structures are rare on the Hudson Bay coast.

Q4- Survival (suitable environment) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Botryllus schlosseri*. Species can survive in other locations (e.g., Gulf of St. Lawrence) under ice cover at sub-zero temperatures. However, these are at the extreme northern extent of its distribution and winters would be shorter and with thinner ice cover than would be expected through some parts of the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: *Botryllus schlosseri* are distributed in cold temperate to subtropical waters and tolerate a wide range of environmental conditions (Fofonoff et al., 2003, Carver et al., 2006a). The species is present north to the Maritimes and Gulf of St Lawrence along the Atlantic coast (Carver et al., 2006a) where waters reach sub-zero temperatures. Its temperature tolerances vary with acclimation and geographical location. Laboratory-raised juvenile colonies were studied using a two-factorial experimental design with five levels of temperature (5, 10, 15, 20, 25 °C) and five levels of salinity (14, 20, 26, 32, 38‰) (Epelbaum et al., 2009a). The colonies tolerated temperatures of (10-25 °C) and salinities of (14-38‰), grew at 10-25 °C and 20-38‰, and attained the largest colony sizes at 15-20 °C and 20-38‰. A minimum temperature tolerance of -1 °C was suggested based on field data (Calder, 1966, Ruiz et al. unpublished data, Whitlatch et al., 1995, https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=159373).

A similar study over a broader range of temperatures (3-28°C) and salinities (16-44 ppt) found 100% adult mortality at 3°C for salinities of 16 and 24 ppt (20% at 38 ppt; 80% at 44 ppt) (Brunetti et al., 1980). In general adult colonies tolerated a wider range of temperature and salinity conditions than did younger colonies.

In Atlantic Canada *B. schlosseri* apparently become dormant or experienced die-back during the winter months, although specific estimates of mortality are not available (Carver et al., 2006a, Deibel et al., 2014). These effects should be magnified in the HB LME where conditions are colder longer, and there is substantial ice scour. Winter temperatures are likely to limit establishment at present. More studies are required to document the species' tolerances and responses to low temperatures and salinities.

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Botryllus schlosseri* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with a maximum sea surface temperature >=10°C, the lowest tested limit at which juvenile colonies could grow. Reproduction is reported to begin at 5°C but temperatures below 5°C have not been tested, so it is uncertain if the life-cycle could be completed at lower temperatures.

Background Information: *Botryllus schlosseri* can reproduce asexually by budding and sexually from fertilized eggs developing into larvae (Fofonoff et al., 2003; and references therein). The zooids are hermaphroditic and each one can produce eggs and sperm. Eggs may be self-fertilized or fertilized by sperm from a nearby animal. Tadpole larvae do not feed and only swim freely for up to 36 hours, settling just prior to metamorphosis. The larvae may settle near another colony and fuse or may grow to form a new colony; most remain within a few meters of the parent colony (Grosberg, 1987). The species' semelparous and iteroparous life history morphs may be a response to a seasonally changing environment (Grosberg, 1988). The former dominate through mid-summer and the latter later in the summer. Asexual growth rates in botryllid colonies are exponential until the onset of sexual maturation, when energy is diverted from budding to egg production (Deibel et al., 2014).

Bud development is influenced by seawater temperature: the zooid life-span, lasts approximately three weeks at 18°C, reduces to 13 days at 26°C, and increases to 65 days at 10°C (Sabbadin, 1955). Laboratory-raised juvenile colonies were studied using a two-factorial experimental design with five levels of temperature (5, 10, 15, 20, 25°C) and five levels of salinity (14, 20, 26, 32, 38‰) (Epelbaum et al., 2009a). The colonies tolerated temperatures of (10-25°C) and salinities of (14-38‰), grew at 10-25°C and 20-38‰, and attained the largest colony sizes at 15-20°C and 20-38‰. Colony growth at 10°C but not 5°C and 20 ppt salinity but not 14 ppt (Epelbaum et al., 2009a) suggests that the lower limits of reproduction are between these values. Asexual and sexual reproduction begin at ca. 5-6°C and ca. 11-13°C, respectively

(Deibel et al., 2014). Populations in more northerly, cooler waters have shorter breeding seasons than those in warmer waters (Carver et al., 2006a, Deibel et al., 2014). They survive stressful environments (e.g., cold) by ceasing sexual reproduction and reducing the rate of budding. More accurate data are needed on reproductive success at low temperatures and salinities.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents could slow population growth in the RA area for *Botryllus schlosseri*. Studies show that population growth may be slowed but not controlled by predation, including by species found in the RA area. Several studies show consistent species-specific effects, including those by the green urchin (*Strongylocentrotus droebachiensis*) which is present in the RA area.

Background Information: Predators may slow population growth but seem unlikely to control it. The patchy distribution of *B. schlosseri* may be attributable to spatial variability in predation pressure on young recruits in the first 3-4 weeks post settlement (Carver et al., 2006a; and references therein; see also Nydam and Stachowicz, 2007). They are eaten by fish (cunner *Tautogolabrous adspersus*) and snails (*Mitrella lunata*, *Anachis* spp.). Adults generally have few predators. While not toxic, they may be unpalatable since extracts of adult *B. schlosseri* deterred feeding activity by shore crab *Carcinus maenas* (Teo and Ryland, 1995). Gastropods, turbellarian flatworms, and nudibranchs such as *Goniodoris castanea* will eat adult *B. schlosseri* in situ, spider crabs eat them in laboratory experiments (Carver et al., 2006a; and references therein). Green urchins, which are present in the HB LME will eat *B. schlosseri* in the laboratory but may prefer other prey in the wild (Deibel et al., 2014).

The epidemiology of "Cup Cell Disease", an infectious disease associated with haplosporidian protists that causes *B. schlosseri* mortality (Moiseeva et al., 2004), is unknown so whether it might slow or control the species' population growth is unknown.

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Botryllus schlosseri* has the capacity for a moderate range of natural dispersal in the RA area. There is some evidence available to suggest a reasonable degree of natural dispersal for asexual buds attached to objects, however the short larval duration limits dispersal at this life stage.

Background Information: Natural dispersal can occur for both asexual buds and tadpole larvae. Under laboratory conditions attached buds can survive 35 days and unattached buds up to 150 days, suggesting that *B. schlosseri* is capable of dispersal via currents or in ballast water over long distances (Rabinowitz and Rinkevich, 2004). The dispersal potential of the larval tadpole stage is limited as it only swims freely for up to 36 hours (Carver et al., 2006a). Hiscock (2008) estimated a larval setting time of <1 d but a dispersal potential of 1-10 km. Other natural dispersal vectors include rafting on eelgrass, algae, or other forms of floating debris (Therriault and Herborg, 2008b); or as epibionts on large crustaceans that move locally or are exported globally (Bernier et al., 2009).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence for wide range dispersal through biofouling, which could occur on vessels that visit the RA area as well as on fishing and recreational vessels that move within the region.

Background Information: Anthropogenic dispersal can occur for both asexual buds and tadpole larvae. Under laboratory conditions attached buds can survive 35 days and unattached buds up to 150 days, suggesting that *B. schlosseri* is capable of dispersal attached to floating debris in ballast water over long distances (Rabinowitz and Rinkevich, 2004, Carver et al., 2006a). Dispersal potential of the larval tadpole stage is limited as it only swims freely for up to 36 hours before settling (Carver et al., 2006a). Hull fouling is another human-mediated pathway for the dispersal of *B. schlosseri* (Lambert and Lambert, 2003, Carver et al. 2006a). Slower towed barges may be more likely potential vectors for transport by hull fouling than faster moving ships and power boats due to the reduced friction on their hull surface. If *B. schlosseri* is introduced, hull fouling on domestic coastal resupply and other boat traffic could spread the species from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Transport of colony fragments in ballast water or on fouled fishing gear is possible, as is ballast water transport of tadpole larvae over smaller distances, but these vectors are less likely to spread the species. There is no aquaculture in the region at present to attract *B. schlosseri* that could be transported with fouled equipment. With the exception of ship-related movements, the importance of other types of movements (e.g., smaller pleasure and fishing vessels) in the RA area is not well known.

Q9- Impact on population (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. Reliable information exists on impacts to populations in invaded locations but findings are equivocal and vary among studies. Effects are largely limited to artificial structures, so strong effects would only be likely in a few areas of the RA area.

Background Information: Rapid growth allows *B. schlosseri* to exploit new environments, potentially displacing native species (Carver et al., 2006a, Dijkstra et al., 2007a, Dijkstra et al., 2007b). They compete for space by overgrowing and smothering existing species, including other fouling species and algae (Pederson et al., 2005). They may produce chemicals that make settlement and growth difficult for other species (Teo and Ryland, 1995, Dijkstra et al., 2007a). They also compete for food with filter feeding taxa such as zooplankton and bivalves (Carver et al., 2006a). The presence of *B. schlosseri* can reduce abundance of some species (mussels, oysters, barnacles) and increase that of others (arborescent bryozoans) (Carver et al., 2006a; and references therein).

Surface fouling by *B. schlosseri* could reduce productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). In James Bay, these plants offer habitat for many marine species and browse for large populations of migratory waterfowl (Stewart and Lockhart, 2005).

Q10- Impact on communities (Score = 2, Certainty = 3

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* would have high impact in few areas or moderate impact in many areas on communities in the RA area. Reliable information exists on impacts to communities in invaded locations, but findings are equivocal and vary among studies. Effects are largely limited to artificial structures, so strong effects would only be likely in a few areas of the RA area.

Background Information: *Botryllus schlosseri* can alter trophic structure of aquatic ecosystems by altering plant, invertebrate, and possibly fish communities (Carver et al., 2006a, Dijkstra et al., 2007a, Dijkstra et al., 2007b). Rapid growth allows them to exploit new environments, potentially displacing native species and disrupting community dynamics. They compete for space by overgrowing and smothering existing species, including other fouling species and algae (Pederson et al., 2005). They also compete for food with filter feeding taxa such as zooplankton and bivalves (Carver et al., 2006a). Surface fouling by *B. schlosseri* could reduce productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). In James Bay, these rich communities offer habitat for many marine species and browse for large populations of migratory waterfowl (Stewart and Lockhart, 2005). Colonial ascidians may produce chemicals that make settlement and growth difficult for other organisms (Teo and Ryland, 1995, Dijkstra et al., 2007a). *Botryllus schlosseri* can be dominant in fouling communities, reducing abundance of some species (mussels, oysters, barnacles) and increasing that of others (arborescent bryozoans) (Carver et al., 2006a; and references therein). In some cases the net impact may be a reduction in community species diversity (Therriault and Herborg, 2008b).

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species can impact habitat structure via overgrowth/competition in invaded areas. These are largely limited to artificial structures, so the species would likely only have strong impacts in a few areas if introduced to the RA area.

Background Information: Introduction of colonial tunicates can reduce habitat availability for other species by covering natural and artificial surfaces (Carver et al., 2006a). Surface fouling by *B. schlosseri* that reduces light transmission could have numerous ecological consequences, including reduced productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). These beds have particular ecological importance for migratory waterfowl in James Bay (Stewart and Lockhart, 2005). Habitat degradation can occur if there is increased fecal deposition or a widespread die-off that leads to an accumulation of decaying biomass on the bottom (Carver et al., 2006a). Under extreme conditions, water quality could be impacted by ammonia excreted by the tunicate colonies. In substantial densities colonial tunicates may increase water clarity by particle removal. This can negatively impact resident filter feeders such as zooplankton and bivalve populations by reducing the amount of food available, but improve light transmission through the water column, enhancing benthic plant production.

Q12- Impact on ecosystem function (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. The species is suggested to be able to impact ecosystem function via overgrowth/competition in invaded areas and perhaps altering water turbidity (and plankton loads) although these are largely limited to artificial structures so strong effects would only be likely in a few areas of the RA area.

Background Information: *Botryllus schlosseri* can alter trophic structure of aquatic ecosystems by altering plant, invertebrate, and possibly fish communities (Carver et al., 2006a, Dijkstra et al., 2007a, Dijkstra et al., 2007b). Rapid growth allows them to exploit new environments, potentially displacing native species and disrupting community dynamics. They compete for space by overgrowing and smothering existing species, including other fouling species and algae (Pederson et al., 2005). Colonial ascidians may produce chemicals that deter predation and make settlement and growth difficult for other organisms (Teo and Ryland, 1995, Dijkstra et al., 2007a). In some cases the net impact may be a reduction in community species diversity (Therriault and Herborg 2008a).

Surface fouling by *B. schlosseri* that reduces light transmission could have numerous ecological consequences, including reduced productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). These beds have particular ecological importance for migratory waterfowl in James Bay (Stewart and Lockhart, 2005), and may currently be relatively free from fouling. Degradation of benthic habitat could occur if there is a substantial increase in tunicate biomass that results in greater deposition of fecal material, especially if widespread mortality of the tunicate colonies contributes additional decaying biomass (Carver et al., 2006a).

In high densities, colonial tunicates may increase water clarity, reducing the amount of food available to native filter feeding taxa such as zooplankton and bivalves but perhaps enhancing benthic plant production (Carver et al., 2006a). They may also affect water quality and nutrients by excreting ammonia. Their tendency to colonize floating substrates and overgrow other species can threaten marine aquaculture operations (Carver et al., 2006a), but there is no aquaculture in the HB LME.

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Botryllus schlosseri* would have low or no impact associated with diseases, parasites or travelers in the RA area. Although there are high rates of parasitism in ascidians generally and evidence of a lethal disease that can be carried by the species, there is little evidence of impacts.

Background Information: Few studies are available on diseases, parasites, and travelers that infect or infest *B. schlosseri* (Carver et al., 2006a, Therriault and Herborg, 2008b). The potential impacts of "Cup Cell Disease", an infectious lethal disease associated with haplosporidian protists (Moiseeva et al., 2004), are unknown.

Saad and Baarakat (2011) suggest that this colonial ascidian provides stable microhabitat for the feeding, shelter, brooding, and reproduction of many marine microorganisms. The found: *Pfiesteria piscicida* (amoeboid and flagellated zoospore stages) and Trichodina myicola; *Cycloporus papillosus*, tunicate crab *Cryptodroma pileifera*, striped barnacle *Balanus amphitrite* (a potentially invasive secies), tubicolous polychaetes including *Sabella crassicornis*, *Sabellaria alveolata*, *Hydroides elegans* and *Terebella lapidaria* on the tunic; *Haplosporidium costale* in the gastric epithelia; *Perkinsus marinus* in epithelial tissues of the branchial chamber (pharynx); *Lankesteria ascidiae* trophozoites in the stomach epithelium or free in the lumen; *Cardiosporidium cionae* sporonts and plasmodia within the haemocytes; *Nephromyces* in the lumen of the digestive system; *Pseudobiceros uniarborensis* in the branchial chamber; *Anisakis simplex* free in the intestine and encapsulated in intestinal epithelium; *Marimermis maritima* in the lumen of the digestive system; and *Thalassonema litoralis* parasitizing the digestive system. Many errant polychaetes were isolated from the tunic and the lumen of the digestive system, including *Nereis virens*, *Nectoneanthes multignatha*, *Namanereis quadraticeps*, *Solomononereis marauensis* and *Eunice kennedy*; the endobiotic amphipods *Leucothoe spinicarpa* and *Anamixis bananarama* were found inside the lumen of the digestive system; *Psammocyclopina georgei* and *Botryllophilus* inside the branchial chamber; the isopod *Aega francoisae* from the cloaca and atrial region; *Mothocya epimerica*, *Anilocra prionuri* and *Bathynomus lowryi* were parasitizing mainly the branchial cavities; and the prosobranch *Littorina littorea* and sea snail *Lamellaria cerebroides* were found embedded in the colonies. The sea slug *Aplysia fasciata* was observed feeding on the tunic of the colonies.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* would have low or no genetic impact on other species in the RA area.

Background Information: The greatest genetic impact would likely occur with other species in the *B. schlosseri* complex (Yund et al., 2015) but the species complex has not been reported from Hudson Strait or Foxe Basin (Stewart and Lockhart, 2005, Goldsmit et al., 2014) (https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* would have low or no impact on at-risk or depleted species in the RA area. Impacts are mainly associated with artificial structures.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay; at-risk fish, bird, and mammal species are unlikely to be affected. Areas of overlap would be few as introduced *B. violaceus* would likely be associated with manmade structures, which are few.

Q16- Impact on fisheries **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Botryllus schlosseri* would have low or no impact on fished species in the RA area. Areas of overlap would be few as introduced *B. violaceus* would likely be associated with manmade structures, which are few. Potential effects related to *B. schlosseri* fouling of eelgrass beds are uncertain.

Background Information: Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. Fouling by colonial tunicates entirely covered some cultured blue mussels in PEI but valve openings were maintained, allowing continued siphoning by the mussels. The fouling had little effect on the growth and production of adult mussels (Arens et al., 2011) but might affect larval settling and smother spat. Commercially harvested anadromous Arctic charr (*Salvelinus alpinus*) are omnivorous and unlikely to be affected, as are harvested marine mammals. Surface fouling by *B. violaceus* could reduce productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). In James Bay, these plants offer habitat for many marine species and browse for large populations of migratory waterfowl that are hunted locally for subsistence and sport (Stewart and Lockhart, 2005).

Q17- Past invasion history **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Botryllus schlosseri* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Colonial ascidians (tunicates), including *B. schlosseri*, are widespread invasive marine invertebrates in shallow-water coastal communities (Lambert and Lambert, 1998, Fofonoff et al., 2003, Dijkstra et al., 2007a). The origin and species status of *B. schlosseri* are uncertain (Fofonoff et al., 2003). Recent genetic studies suggest it is a complex of at least 3 previously unrecognized and probably reproductively isolated cryptic species--perhaps 5 (clades A-E) (Bock et al., 2012). Only one of these (clade A) has a widespread global distribution, occurring in the northeastern and northwestern Atlantic and northeastern Pacific. The other clades occur mostly in northern Europe. Clade A is comprised of at least 5 subclades, at least one of which is native to Atlantic North America and has been introduced to sites in the northeastern Atlantic and Mediterranean (Yund et al., 2015). *B. schlosseri* has been introduced and established along the west coast of North America, west and southeast coasts of South America and Africa, east coast of India, and around Australia and New Zealand (Fofonoff et al., 2003). Introductions along the Labrador coast have failed to establish.

Ciona intestinalis (Linnaeus, 1767)

Phylum: Chordata

Class: Ascidiacea

Order: Phlebobranchia

Family: Cionidae

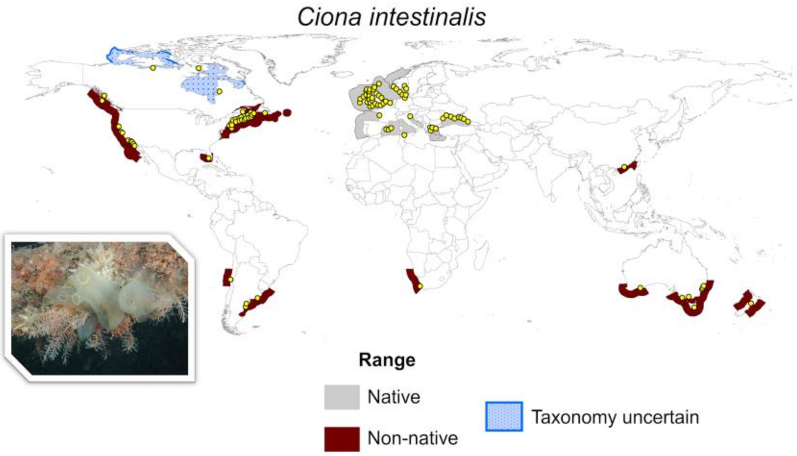

Figure 17: Ecoregions where Ciona intestinalis is distributed: native (grey) and non-native (dark red) regions. Ecoregion where the status is taxonomically uncertain has been marked in blue. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Ciona intestinalis occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.wt6wnw, 27 June 2017), Wacasey et al. (1980), Atkinson and Wacasey (1989a, 1989b, 1989c), Madariaga et al. (2014). Note: These are the likely native and non-native distributions based on the literature, but the species is considered cryptogenic to some regions. Picture of C. intestinalis modified from https://invasions.si.edu/nemesis/species_summary/159113.

CMIST scores for *C. intestinalis:*

Mean adjusted Likelihood of Invasion: 2.22

Mean adjusted Impact of Invasion: 1.93

Mean adjusted Overall CMIST score: 4.29

Q1- Present status in the area (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Ciona intestinalis* has been observed but is not reported as established in the RA area. There is evidence that the species has been found at several Arctic locations including the RA area. However, it has a complex taxonomy and there are doubts as to whether it is native or introduced in the Arctic. Survey effort is low relative to the size of the RA area.

Background Information: Ciona intestinalis has been reported from Richmond Gulf, Hudson Bay, Repulse Bay, Frobisher Bay and the Bathurst Inlet-Dease Strait area (Wacasey et al., 1980, Atkinson and Wacasey, 1989b, c, a). Carver et al. (2006b) suggested that *C. intestinalis* in the Canadian Arctic may belong to one of the cold-water subspecies in the *C. intestinalis* complex (f. *gelatinosa* or f. *longissima*) rather than *C. intestinalis* f. *typica*, but these subspecies may in fact be true species (Sanamyan and Sanamyan, 2007). It has not been detected in any recent surveys in the RA area (Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016).

Ciona intestinalis f. *typica* is established along the Canadian Atlantic coast (NS, PEI, NFLD) (Carver et al., 2006b, Therriault and Herborg, 2008a, Zhan et al., 2012, Sargent et al., 2013). It has not been reported along the Canadian Pacific coast but has been reported in Puget Sound, WA.

Ciona intestinalis (formerly *C. intestinalis* type B) is likely native to the NE Atlantic but non-native or cryptogenic in the NW Atlantic (e.g. Carver et al., 2006b, Zhan et al., 2012) and also occurs in the Bohai and Yellow Seas, China (Zhan et al., 2010). *Ciona robusta* (formerly *C. intestinalis* type A) is likely native to the NW Pacific and introduced in the Atlantic, Mediterranean Sea, Oceania, and North and South Pacific oceans (Bouchemousse et al., 2016a). The two species are sympatric in the western English Channel and south of Brittany, where *C. robusta* has likely been introduced into the native range of *C. intestinalis* (Nydam and Harrison, 2011, Bouchemousse et al., 2016b). *Ciona intestinalis* and *C. robusta* have shared phylogeny and life-history traits, and similar environmental requirements (Bouchemousse et al., 2016a).

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Ciona intestinalis* could arrive frequently in low numbers or infrequently in high numbers in the RA area. The species is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Propagule pressure is unknown but *C. intestinalis* is present at ports directly connected to Churchill, Deception Bay, and Iqaluit by shipping (Chan et al., 2012; see also Bouchemousse et al., 2016a). Hull fouling is the most likely vector for introduction and subsequent dispersal (Therriault and Herborg, 2008a).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale : Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat conditions for *Ciona intestinalis*. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the habitat classification.

Background Information: Except during a few days spent in the water column as mobile larvae, *C. intestinalis* live in the subtidal zone attached to natural substrates such as rocks, eelgrass and kelp or to anthropogenic substrates such as wood, metal or concrete docks, pilings and aquaculture gear (Dybern, 1965, McDonald, 2004, Sargent et al., 2013). They are often observed in shallow coastal waters but occur to a depth of ca. 200 (Therriault and Herborg, 2008a) to 500 m (Brunel et al., 1998).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Ciona intestinalis*. It can tolerate cold environments and those temperatures could be found in a moderate extension of the RA area. The species can survive under ice cover at sub-zero temperatures in other locations (e.g., Gulf of St. Lawrence). Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Ciona intestinalis is a cold-water (or temperate) ascidian (Carver et al., 2006b). Its survival and growth are directly dependent on water temperature. *Ciona intestinalis* f. *typica* is established along the Canadian Atlantic coast (NS, PEI, NFLD) (Carver et al., 2006b, Therriault and Herborg, 2008a, Zhan et al., 2012, Sargent et al., 2013) extending north to areas with ice cover and sub zero winter water temperatures as found in the assessment area. Populations tolerate temperatures from -1 to 30°C (Dybern, 1965), grow faster at higher temperatures (ca. 18°C) than at low ones (0°C), and show heat stress at about 24°C (Dybern, 1965, Petersen and Riisgård, 1992, Carver et al., 2003, Vercaemer et al., 2011, Madariaga et al., 2014). In 2006 the temperature at 3 m depth in PEI waters with *C. intestinalis* ranged from -1.5°C under ice cover to 21°C in August; salinity from 15 PSU in the fall to 30.1 PSU in the spring; and dissolved oxygen from 4.9 to 10.1 mg/L, with the highest levels being observed in the spring (Carver et al., 2003). Over a 12-week lab experiment all *C. intestinalis* exposed to 24°C died, regardless of salinity (range 20-35 PSU) (Vercaemer et al., 2011). *Ciona intestinalis* can tolerate salinities of 12‰ to 40‰ (euryhaline) (Carver et al., 2006b). They are usually found in full salinity seawater and may be able to withstand short-term salinity fluctuations (Therriault and Herborg, 2008a). Over a 12-week lab experiment all *C. intestinalis* exposed to 20 PSU died, regardless of temperature (range 8-24°C) (Vercaemer et al., 2011). Adult *C. intestinalis* may be more tolerant of lower temperatures (<0 cf. ca. 3°C) and salinities (9 cf. 21-25) than *C. robusta* (Fofonoff et al., 2003; and references therein). Further evaluation is needed of the species' tolerance to low temperature (<0°C) at a range of salinities. Specimens found in the Arctic (including within the assessment area) are clearly capable of surviving in the assessment area, but may belong to one of the cold-water subspecies in the *C. intestinalis* complex (f. *gelatinosa* or f. *longissima*) rather than *C. intestinalis* f. *typica* which is found on Atlantic coast.

A thin layer of sediment deters larval settlement by *C. intestinalis*, and clogs the filtration apparatus of settlers (Lowen et al., 2016a).

In California, winter rains resulted in massive die offs (Lambert and Lambert, 1998). Dilution of HB LME surface waters by spring runoff and ice melt might help control *C. intestinalis* on shallow man-made structures.

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Ciona intestinalis* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with a maximum sea surface temperature >=6ºC, the lowest tested limit for reproduction/recruitment. Testing has not been conducted at temperatures below 6ºC, only observations of recruitment patterns, so it is uncertain if reproduction could be completed at lower temperatures.

Background Information: In PEI, ripe eggs were present in the ovary of *C. intestinalis* from November (7 °C) through January (1 °C) (Carver et al., 2003). Gametes remaining in the gonads were non-viable by early December (Harris et al., 2017). There were signs of egg resorption in February and March (<3 to -1 °C), with gametogenesis in March to May at 4 to 8 °C, and the production of competent gametes from mid-May onward when water temperatures exceeded 8 °C (Carver et al., 2003). The first recruitment was observed when water temperatures rose above 8 °C and ceased when they fell below 8 °C (Ramsay et al., 2009). The recruitment levels increased with increasing temperature through the summer season. In Atlantic Canada, the first generation reaches maturity within 8-10 weeks and then gives rise to a second generation within the same year (Carver et al., 2006b).

Sexual reproduction by *C. intestinalis* is constrained to temperatures of > 6 ° to 8 ºC throughout its known range (Carver et al., 2003, Lowen et al., 2016b). Zygotes can survive at <6 to 23 °C and larvae 6 to 24 °C (Dybern, 1965) but perform better at intermediate temperatures (Rius et al., 2014). Vercaemer et al. (2011) found recruitment continued at temperatures down to 6-8 ºC and Harris et al. (2017) found in situ recruitment started in early June when temperatures were ~ 8 ºC and continued through November -December when temperatures declined to 8-10 ºC.

Water temperature can alter embryonic and larval development of both *C. intestinalis* and *C. robusta* (Sato et al., 2015). Laboratory testing found *C. robusta* was better able to cope with heat stress (27 °C for 1 hr or 24 °C overnight). In hybrids this temperature buffering capacity was maternally inherited. The effects of cold stress were not tested.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents could slow population growth in the RA area for *Ciona intestinalis*. Several studies in other areas show species-specific effects from species that also occur in the RA area and could slow population growth but not necessarily restrict it severely.

Background Information: Outside its native range, competition and/or predation pressure can constrain the distribution of *C. intestinalis* to artificial surfaces (Carver et al., 2003, Carver et al., 2006b). Whether this would occur in the HB LME is unknown. During field experiments in New England, ascidians, including *C. intestinalis*, were rare or absent where predators were present (Osman and Whitlatch, 2004). Small predatory gastropods (*Anachis* spp. and *Mitrella lunata*) were effective predators of recently settled juveniles. None of the *C. intestinalis* recruits survived on piling surfaces exposed to predation by large invertebrates and fishes. Heavy grazing of the sea star *Asterias rubens* on adult *C. intestinalis* has been reported from eelgrass beds in Norway (Gulliksen and Skjæveland, 1973) and Sweden (Petersen and Svane, 1995). In the laboratory, green crab (*Carcinus maenas*) and rock crab (*Cancer irroratus*) fed on juvenile (20-50 mm) and adult (50-100 mm) *C. intestinalis* (Carver et al., 2003). Surface and bottom feeding fishes that may play a role in controlling the population dynamics of *C. intestinalis* include dab (*Limanda limanda*), plaice (*Pleuronectes platessa*), Atlantic cod (*Gadus morhua*) (Gulliksen and Skjæveland, 1973), cunner (*Tautogolabrous adspersus*) (Osman and Whitlatch, 2004), and three-spined stickleback (*Gasterosteus aculeatus*) (Petersen and Svane, 1995). Several of these species (cod and stickleback) are locally present in the HB LME (Stewart and Lockhart, 2005).

From NEMESIS: In Prince Edward Island, Canada, high densities of caprellids (*C. mutica* and *C. linearis*) were shown to inhibit the settlement of *C. intestinalis* (Collin and Johnson, 2014). Since the tunicate appears to be locally invasive, and interfering with mussel aquaculture, there may be a potential for biocontrol by encouraging caprellid populations (Collin and Johnson, 2014) (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=159113).

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Ciona intestinalis* has the capacity for a moderate range of natural dispersal in the RA area. There is some evidence to suggest natural dispersal of sessile adults attached to objects. The species' longer larval duration (4-10 days) relative to other tunicates (1-2 days) would facilitate dispersal at this life stage compared to most other tunicates.

Background Information: The free-swimming larval stage of *C. intestinalis* only lasts 4-5 days at 10-12 °C or 24-36 hours at 18-20 °C (Dybern, 1965). This short duration limits larval dispersal, which depends on swimming speed and direction, duration of the tadpole phase, and local currents (Therriault and Herborg, 2008a).

From Carver et al. (2006b): "Laboratory studies generally report that the duration of the period from egg release to larval settlement does not exceed 1 week at temperatures of 10-20 °C (Dybern, 1965, Cirino, 2002). This may be extended in the field if egg fertilization is delayed and/or larvae cannot find suitable substrate for settlement; Jackson (2008) suggested a maximum of 10 days for British waters, but few field data are available. Most range extensions likely occur after settling when sessile adults hitchhike on natural or artificial substrates. Other tunicate species have been observed to raft on natural floating substrates such as eelgrass or macroalgae (Worcester, 1994). In a recent tunicate survey on the South shore of Nova Scotia, *C. intestinalis* was repeatedly found rafting on clumps of the fleece alga *Codium fragile* (Vercaemer, pers. Obs.)."

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* has the capacity for a wide range of anthropogenic dispersal in the RA area. There seems to be much literature suggesting wide range dispersal through biofouling which could occur on vessels known to move within the RA area as well as on fishing and recreational vessels likely to move within the same region.

Background Information: Hull fouling of slow-moving vessels and recreational watercraft is likely responsible for regional dispersal within many coastal areas (Lambert and Lambert, 1998, Carver et al. 2006b). Hull fouling on domestic coastal resupply and other boat traffic could spread *C. intestinalis* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Transport of colony fragments in ballast water or on fouled fishing gear is possible, as is ballast water transport of tadpole larvae over smaller distances, but these vectors are less likely to spread the species (Carver et al., 2006b). There is no aquaculture in the region at present to attract *C. intestinalis* that could be transported with fouled equipment. With the exception of ship-related movements, the importance of other types of movements (e.g., smaller pleasure and fishing vessels) in HBC is not well known.

Given the species' rapid spread through the St. Lawrence, particularly in marinas, it is clear that small craft are an important and efficient vector for this animal.

Q9- Impact on population (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. The species is known to impact populations in invaded areas. Because these populations are largely limited to artificial structures, strong effects should be localized to a few small areas of the RA area.

Background Information: Species such as *C. intestinalis* that settle early in the season may outcompete species that settle later simply because their larger size allows for greater energetic reserves and/or greater feeding capacity (Carver et al., 2006b). *Ciona intestinalis* competition for phytoplankton may lower the carrying capacity of bays in which blue mussels (*Mytilus edulis*) are cultured (Comeau et al., 2015). In many areas where *C. intestinalis* has been introduced, predation has limited population growth, except on manmade structures that serve as refuges (Lambert and Lambert, 1998). Man-made structures are rare in the HB LME, which could limit the ability of *C. intestinalis* to establish and spread if predation pressure on *C. intestinalis* in natural habitats is significant, thereby limiting impacts on indigenous species.

Lambert and Lambert (1998) noted that since *C. intestinalis* appeared in southern California in 1917, there has been a steady decline in the abundance of native ascidian species which previously dominated, an observation that is more apparent on floating structures where predators are absent or have less impact on biofouling communities.

Q10- Impact on communities (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* would have high impact in few areas or moderate impact in many areas on communities in the RA area. The species is known to impact communities in invaded areas, although these are largely limited to artificial structures, so strong effects would only be likely in a few areas of the RA area.

Background Information: Dense aggregations can change species richness and community composition, with a moderate impact on biodiversity (Therriault and Herborg, 2008a). Since *C. intestinalis* was first reported in southern California in 1917, there has been a steady decline in the abundance of native ascidian species which previously dominated (Lambert and Lambert, 1998). This decline is most apparent on floating structures where predators are absent or have less impact on biofouling communities. These structures may serve *C. intestinalis* as a refuge from predation, enabling it to flourish. The rarity of these structures in the HB LME could limit the ability of *C. intestinalis* to establish and spread if predation pressure on *C. intestinalis* in natural habitats is significant, thereby limiting community impacts.

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species is known to impact pelagic and benthic habitats in invaded areas, although these are largely limited to artificial structures, so strong effects would only be likely in a few areas of the RA area.

Background Information: Ciona intestinalis forms dense aggregations and takes up space that could be used as habitat for other species (Therriault and Herborg, 2008a). These aggregations can negatively impact habitat structure and potentially food web and trophic structure of aquatic ecosystems by inducing changes in plant, invertebrate and possibly fish communities (Carver et al., 2006b). In high densities, *C. intestinalis* and *Styela clava* can reduce plankton loads in enclosed embayments used for aquaculture (Comeau et al., 2015), and substantially increase biodeposition of fecal pellets and ascidian carcasses to the bottom sediment (McKindsey et al., 2009, Qi et al., 2015). However, there are no shellfish aquaculture operations in the HB LME. In other areas predation has limited population growth, except on manmade structures that serve as refuges (Lambert and Lambert, 1998). Man-made structures are rare in the HB LME, which could limit the ability of *C. intestinalis* to establish and spread if predation pressure on *C. intestinalis* in natural habitats is significant, thereby limiting impacts on natural habitats.

Q12- Impact on ecosystem function (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. The species is known to impact pelagic and benthic habitats in invaded areas, although these are largely limited to artificial structures so strong effects would only be likely in a few areas of the RA area.

Background Information: Dense aggregations can decrease water circulation, limiting oxygen and food (Therriault et al., 2008b). In high densities, *C. intestinalis* and *Styela clava* can reduce plankton loads in enclosed embayments used for aquaculture (Comeau et al., 2015), and substantially increase biodeposition of fecal pellets and ascidian carcasses to the bottom sediment (McKindsey et al., 2009, Qi et al., 2015). However, there are no shellfish aquaculture operations in the HB LME. In other areas predation has limited population growth, except on manmade structures that serve as refuges (Lambert and Lambert, 1998). Man-made structures are rare in the HB LME, which could limit the ability of *C. intestinalis* to establish and spread if predation pressure on *C. intestinalis* in natural habitats is significant, thereby limiting impacts on natural habitats.

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Ciona intestinalis* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. There is published literature on potential effects.

Background Information: Several parasitic diseases have been documented in *C. intestinalis.* Those in Tasmania (presumably species “A” now *C. robusta*) can carry *Neoparamoeba pemaquidensis* in their branchial sacs (Tan et al., 2002). This parasite may be responsible for Amoebic Gill Disease (AGD) which is a concern for cultured salmon, particularly Atlantic salmon (*Salmo salar*), and affects numerous salmonids and other marine fishes such as the turbot

(*Scophthalmus maximus*) (Munday et al., 2001, Ruane and Jones, 2013). *Neoparamoeba perurans* also causes AGD (Young et al., 2008). While no reports of this species were found from *C. intestinalis*, these *Neoparamoeba* species were only recently separated, so it may also be carried on *C. intestinalis*. *Neoparamoeba pemaquidensis* has been reported at temperatures as low as 9 °C but is sensitive to exposure to fresh water. No information was found on the susceptibility of Arctic char (*Salvelinus alpinus*) in the HB LME to AGD, but their anadromous lifestyle, which ensures they spend most of the year in fresh water, should limit risk of harm to these fish stocks. *Ciona intestinalis* and other ascidians may also transport viable cells and cysts of toxic phytoplankton (Rosa et al., 2013).

From Mita et al. (2012): “Here we report a disease of *C. intestinalis* observed in an inland culturing system. The disease, called ‘long feces syndrome,’ is expressed in affected animals by the following characteristic symptoms of the digestive system: (1) excretion of long and thin feces, (2) pale color of the stomach, and (3) congestion of the digestive tube by digested material. Severely diseased animals usually die within a week after the first symptoms occur, implying a high risk of this disease for ascidian culturing systems. The digestive tubes of the diseased animals are occupied by the gregarine apicomplexan parasite *Lankesteria ascidiae*, suggesting that large-scale infection by this parasite is the cause of long feces syndrome.”

Q14- Genetic impact (Score = 3, Certainty = 1)

Score Rationale: Little to no reliable information suggests *Ciona intestinalis* would have high genetic impact in many areas on other species in the RA area. There are co-occurring native tunicates of the same genus in the RA area. There is evidence of hybridization within the species complex, but sometimes not, depending on which species. Taxonomy of Arctic species is also uncertain, but published information indicates it is most likely a distinct species within this complex.

Background Information: The genetic effects of introduction will depend upon whether the indigenous species identified as *C. intestinalis* and the introduced species interbreed to produce fertile offspring. *Ciona intestinalis* (formerly *C. intestinalis* species “B”) belongs to a species complex that includes *Ciona robusta* (formerly *C. intestinalis* species “A”) and several other species, subspecies, or forms (Fofonoff et al., 2003). Populations of *C. intestinalis* and *C. robusta* overlap in the English Channel but seldom hybridize and the hybrids are infertile (Caputi et al., 2007, Sato et al., 2012, Sato et al., 2014, Sato et al., 2015). The taxonomic identity *of C. intestinalis* reported from the HB LME is uncertain so the genetic effects of *C. intestinalis* introduction are also uncertain.

From Malfant et al. (2017): “The introduction of the warm-temperate tunicate *Ciona robusta*, native to Asia, into the European native range of its cold-temperate congener *Ciona intestinalis*. Despite an old divergence (ca. 4 *Mya*), the two species lack reproductive isolation with fertile F1-hybrids produced in the laboratory. However, a recent population genomic study showed that hybrids are extremely rare in the wild and back-crossed individuals are absent, suggesting the existence of reproductive barriers between the two species in nature. Post-zygotic mechanisms like selection against hybrids by the environment were thus hypothesized. Because the two species have only been recently reclassified, data on their respective ecology and their hybrids are lacking. To investigate these environmental effects, we carried out 21 experimental crosses and examined survival and individual growth rate of ca. 1900 F1-juveniles produced with intra- and inter-specific crosses. Four treatments combining two temperatures (12 °C and 17 °C) and two salinity levels (25 and 35), representative of conditions found in nature, were applied to one-week old juveniles. After 28 days, a high survival rate was observed in the three F1 categories, with no difference between hybrids and parental species. As expected based on biogeography studies and field monitoring of the two species, the results showed that *C. robusta* performed particularly well in warm temperature with standard salinity condition. F1-

hybrids did not show outbreeding depression, with growth rates similar to or higher than F1-*C. intestinalis*. Altogether the two species performed well under the tested environmental conditions, and post-zygotic ecological effects alone are unlikely to be effective against hybridization between the native and non-native species."

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* would have low or no impact on at-risk or depleted species in the RA area.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay; at-risk fish, bird, and mammal species are unlikely to be affected. Areas of overlap would be few as introduced *C. intestinalis* would likely be associated with manmade structures, which are few.

Q16- Impact on fisheries (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* would have low or no impact on fished species in the RA area. Areas of overlap would be few as introduced *C. intestinalis* would likely be associated with manmade structures, which are few.

Background Information: Shellfish are not cultivated in the HB LME. Natural populations of blue mussel (*Mytilus edulis*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, and occasionally for subsistence by Hudson Bay residents (Stewart and Lockhart, 2005). Damage to mussel populations in the Hudson Bay LME could also affect Hudson Bay eider (*Somateria mollissima sedentaria*) that overwinter in polynyas in the Belcher Islands, where they eat mussels and are hunted for subsistence.

Ciona intestinalis may carry an amoebic parasite that affects fishes of the genera *Salmo* and *Oncorhynchus* (Munday et al., 2001, Ruane and Jones, 2013). Whether it would cause disease mortality among anadromous Arctic char (*Salvelinus alpinus*) or brook trout (*Salvelinus fontinalis*) is unknown. Both species are harvested for subsistence and char also for commercial sale (Stewart and Lockhart, 2005).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Ciona intestinalis* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Ciona intestinalis (formerly *C. intestinalis* type B) is likely native to the northeast Atlantic but non-native or cryptogenic in the northwest Atlantic (e.g. Carver et al., 2006b, Zhan et al., 2012) and also occurs in the Bohai and Yellow Seas in China (Zhan et al., 2010). *Ciona robusta* (formerly *C. intestinalis* type A) is likely native to the northwest Pacific and introduced in the Atlantic, Mediterranean Sea, Oceania, and North and South Pacific oceans (Bouchemousse et al., 2016a). The two species are sympatric in the western English Channel and south of Brittany, where *C. robusta* has likely been introduced into the native range of *C. intestinalis* (Nydam and Harrison, 2011, Bouchemousse et al., 2016b). *Ciona intestinalis* and *C. robusta* have shared phylogeny and life-history traits, and similar environmental requirements (Bouchemousse et al., 2016a). Both are sessile as adults, with a short life cycle involving broadcast spawning for external fertilization, and a non-feeding larva with a short planktonic phase (Bouchemousse et al., 2016b). The relationships of these taxa to *C. intestinalis* f. *longissima*, and *C. intestinalis* f. *gelatinosa* that have been reported from Arctic waters (Dybern, 1965), and may be represented in the HB LME (Carver et al., 2006b), is uncertain and requires further study to support invasion risk and impact predictions.

Older literature (ca. pre-2010) does not always distinguish between *C. intestinalis* and *C. robusta* so environmental tolerances, habitat preferences, and impacts should be interpreted with care.

Predicted increases in ambient sea temperature are likely to result in earlier sexual maturation and extend the reproductive window for *C. intestinalis* in Atlantic Canada (Harris et al., 2017) and also in the HB LME.

Molgula manhattensis **(De Kay, 1843)**

Phylum: Chordata
Class: Ascidiacea
Order: Stolidobranchia
Family: Molgulidae

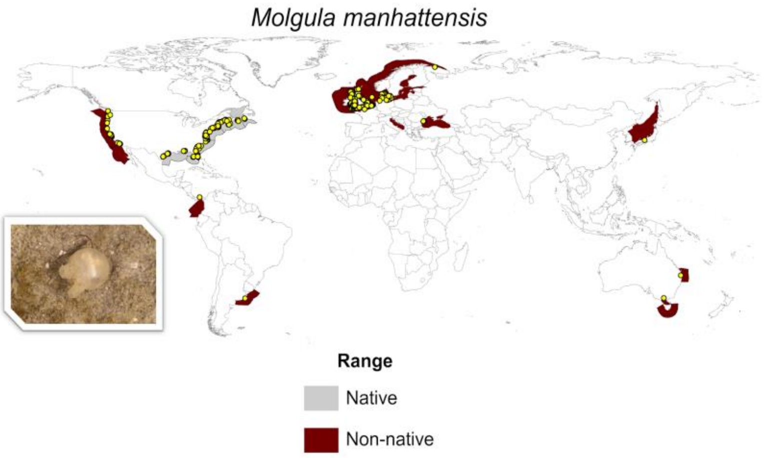

Figure 18: Ecoregions where Molgula manhattensis is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Molgula manhattensis occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.3708ta, 27 June 2017), Preda et al. (2012), Haydar et al. (2011). Picture of M. manhattensis modified from https://invasions.si.edu/nemesis/species_summary/159557.

CMIST scores for *M. manhattensis:*

Mean adjusted Likelihood of Invasion: 1.90

Mean adjusted Impact of Invasion: 1.89

Mean adjusted Overall CMIST score: 3.59

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Molgula manhattensis* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found of the species being present in Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). The introduced range of *M. manhattensis* on the BC coast extends from the Strait of Georgia to Prince Rupert (Levings et al., 2002, Clarke Murray et al., 2011).

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Molgula manhattensis* could arrive frequently in low numbers or infrequently in high numbers in the RA area. The species is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Propagule pressure is unknown but *M. manhattensis* is present at ports in Europe (Gollasch et al., 2009) and along the Atlantic coast of North America (Fofonoff et al., 2003), where it could foul the hulls or be entrained in ballast water of ships and transported to Churchill.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Molgula manhattensis*. There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support habitat classification.

Background Information: Molgula manhattensis can be found from the intertidal zone to ca. 90 m depth, prefer protected marine waters and estuaries, and are very tolerant of pollution (http://wdfw.wa.gov/ais/molgula_manhattensis/; Lambert and Lambert, 1998, 2003, Zvyagintsev and Ke, 2003, Carman et al., 2007). They attach to a variety of natural surfaces including gravel, sand, seaweed, and shellfish and to anthropogenic structures such as marinas, docks, piers, floats and buoys, ropes and wires, ship and boat hulls, and shellfish aquaculture operations.

Q4- Survival (suitable environment) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Molgula manhattensis*. It can tolerate cold environments and those temperatures could be found in a moderate extension of the RA area. The species can survive under ice cover at sub-zero temperatures in other locations (e.g., Gulf of St. Lawrence). However, *M. manhattensis* at these locations are at the extreme northern extent of their distribution and winters would be shorter and ice cover thinner than in much of the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: It is tolerant of a wide range of temperatures, salinities and pollution levels and is found on both natural and artificial substrates (Fofonoff et al., 2003; and references therein). In the Patuxent River estuary, Maryland *M. manhattensis* occurs at temperatures ranging from ca. 0 to 28 °C and salinities from ca. 6 to 20 ppt (Cory, 1967). Downstream, in Chesapeake Bay, it occurs at salinities >10 ppt (Wass, 1972). The species occupies waters from the Gulf of Mexico north to the Gulf of St Lawrence, so it may be able to withstand a broader range of water temperatures as it occurs in areas where waters are ice covered and reach sub-zero. It has been reported in the field at salinities up to 35 ppt (Fofonoff et al., 2003). It is one of the few ascidian species that can tolerate low salinity (Cohen, 1995, Lambert and Lambert, 1998, Zvyagintsev and Ke, 2003), which could be an important consideration in the HB LME where spring melt adds large volumes of fresh water from runoff and ice melt to the surface waters (Stewart and Lockhart, 2005). *Molgula manhattensis* is abundant in Massachusetts harbours with poor water quality (i.e., high levels of nitrogen, low oxygen (hypoxia), no eelgrass beds, poor water clarity, high suspended sediment) as well as in fair and good water quality (Zvyagintsev and Ke, 2003, Carman et al., 2007). It is also relatively tolerant to hypoxia in Chesapeake Bay (Sagasti et al., 2000, Sagasti et al., 2003).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest reproductive requirements are sometimes available for *Molgula manhattensis* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with maximum sea surface temperature >=10 °C, the lowest tested limit at which reproduction is reported to occur. Testing has not been conducted at temperatures below 10 °C, so it is uncertain if the life cycle could be completed at lower temperatures.

Background Information: Molgula manhattensis is hermaphroditic and capable of self-fertilization although success of embryonic development can be lower than for cross fertilization (Morgan, 1942). Reproduction begins when water temperature reaches about 10 °C (Lützen, 1967; https://www.nobanis.org/globalassets/speciesinfo/m/molgula-manhattensis/molgula-manhattensis.pdf). Eggs and sperm are broadcast spawned in the water column (http://wdfw.wa.gov/ais/molgula_manhattensis/). Fertilized eggs develop into free-swimming tadpole larvae in about 24 h (Grave, 1933). No clear data were found on the duration of the larval stage. Jensen (2010; NOBANIS Fact sheet) suggested it lasts only a few days based on Saffo and Davis (1982), but that information was not found in the paper. When suitable habitat is found the larvae settle, attach, and metamorphose into adults. The free swimming larval stage may be by-passed and metamorphosis completed *in situ* (Morgan, 1942). Under favorable conditions sexual maturity is reached in 3 weeks, although fertility increases after one month (Grave, 1933). In southern Russia, settlement of juveniles was observed from 13 and 22 °C (Zvyagintsev and Ke, 2003).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Molgula manhattensis.* Several studies show species-specific effects; none of these species is known to occur in the RA area, however similar types of species that could exert predation pressure may be found there.

Background Information: During field experiments in New England the survival of *M. manhattensis* varied inversely (weakly) with predator abundance (Osman and Whitlatch, 2004). As recruits increased in size they escaped predation by the very small predaceous gastropod *Mitrella lunata* only to suffer increased predation by the larger *Anachis lafresnayi*. Under experimental conditions both gastropod species reduced the abundance of *M. manhattensis* settlers (Osman and Whitlatch, 1995). Some fish (e.g., cunner *Tautogolabrus adspersus*) are strongly attracted to and efficient predators of juvenile *M. manhattensis* in the field (Osman et al., 1990, Osman and Whitlatch, 2004). They also eat adults but at lower rates. Green crabs *Carcinus maenas* and spider crabs *Libinia dubia* eat *M. manhattensis* under both field and laboratory conditions (Carman and Grunden, 2019).

Q7- Spread (potential natural dispersal) (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Molgula manhattensis* has the capacity for a very limited range of natural dispersal in the RA area. Certainty about the species' natural dispersal is low.

Background Information: Natural dispersal of the planktonic larvae will be limited by the duration of the larval stage, which is uncertain for *M. manhattensis* and typically short and variable within ascidian species (Svane and Young, 1989, Haydar et al., 2011). Larvae may also be more likely to settle near members of their own species (Stachowicz et al., 1999). Most range extensions likely occur after settling when sessile adults hitchhike on natural or artificial substrates (Haydar et al. 2011). *Molgula manhattensis* sometimes attaches to large crustaceans, such as the green crab *Carcinus maenas* (Crothers, 1968, Bernier et al., 2009).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Molgula manhattensis* has the capacity for a wide range of anthropogenic dispersal in the RA area. There seems to be much general literature suggesting wide range dispersal of tunicates through biofouling which could occur on vessels that move within the RA area, as well as on fishing and recreational vessels likely to move within the region.

Background Information: Hull fouling on domestic coastal resupply and other boat traffic could spread *M. manhattensis* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Clarke Murray et al., 2011, Chan et al., 2012). Transport of adults on fouled fishing gear is possible, as is ballast water transport of tadpole larvae over smaller distances, but these vectors are less likely to spread the species. There is no aquaculture in the region at present to attract *M. manhattensis* that could be transported with fouled equipment. With the exception of ship-related movements, the importance of other types of movements (e.g., smaller pleasure and fishing vessels) in the RA area not well known.

Q9- Impact on population (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Molgula manhattensis* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. The species is known to impact many populations in invaded areas. Although effects are variable among studies, they may be more wide-spread as this tunicate is not as reliant on artificial structure as others.

Background Information: Native *M. manhattensis* in Chesapeake Bay are capable of rapidly settling on and overgrowing most other fouling community organisms (Andrews, 1953, Calder, 1966, Otsuka and Dauer, 1982). The species frequently attains 100% cover on settling plates (Otsuka and Dauer, 1982). Introduced populations can become extremely dense. In San Francisco Bay they have clogged bottom trawls (Ganssle, 1966). When another tunicate, *Ciona intestinalis* was removed in experimental trials, *M. manhattensis* covered up to 92% of fouling plates (Blum et al., 2007). Complete cover of *M. manhattensis* was observed at some sites and times in Newport and Alamitos Bays, California, but was absent from most sites (Lambert and Lambert, 2003).

Q10- Impact on communities (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Molgula manhattensis* would have high impact in few areas or moderate impact in many areas on communities in the RA area. The species is known to impact many communities in invaded areas. Although effects are variable among studies, they may be more wide-spread as this tunicate is not as reliant on artificial structure as others.

Background Information: Once established, *M. manhattensis* can displace native fouling organisms by out-competing them for food and space, and potentially by consuming the spawn or larvae of other marine species. Native *M. manhattensis* dominated the fouling communities that developed on test panels at Lynnhaven, VA, with and without exposure to predators (Otsuka and Dauer, 1982). It was able to invade and rapidly overgrow all resident species except *Balanus eburneus*. At Bear's Bluff, South Carolina, *Balanus eburneus* was highly dominant before being smothered by *M. manhattensis* (Buchanan, 1975). The presence of *M. manhattensis* and the hydroid *Tubularia crocea* on settling plates may facilitate mussel recruitment (Dean and Hurd, 1980).

Q11- Impact on habitats (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Molgula manhattensis* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. Although the species does add physical structure, it is uncertain how much it impacts the organisms associated with this structure. The species may have moderate wide-spread effects as it is less reliant on artificial structures than other ascidians.

Background Information: The rapid growth of *M. manhattensis* enables it to quickly cover fouled surfaces with multiple layers of tunicates ~10-20 mm deep (Fofonoff et al., 2003). This is likely to inhibit settlement of many other organisms, although a variety of taxa settle on its tunic (e.g., hydroids, *Polydora* spp., *Corophium* spp., bryozoans, ascidians) (Otsuka and Dauer, 1982, Osman and Whitlatch, 1995). The presence of *M. manhattensis* on settlement plates did not affect recruitment on adjacent bare surfaces (Osman and Whitlatch, 1995). When *M. manhattensis* accumulations become excessive, slough-offs of large individuals can rapidly reduce the species' coverage enabling other species to settle seasonally or in succession (Otsuka and Dauer, 1982). *Molgula manhattensis* has a high production rate of solids per unit tissue weight relative to other suspension feeders such as oysters and could influence biodeposition, transport, and composition of suspended sediments in estuaries (Haven and Morales-Alamo, 1966).

Q12- Impact on ecosystem function (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Molgula manhattensis* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. There are possible effects on ecosystem function, but there is high uncertainty given conflicting reports and limited available information.

Background Information: Washington State, USA on the Pacific coast has implemented a program to manage invasive tunicates in ports and marinas. *Molgula manhattensis* has been given low priority because it has not demonstrated a high invasive threat (LeClair et al., 2009). A panel of fifteen experts believed it had the lowest relative risk among 7 nonindigenous tunicates based primarily on perceived and demonstrated threats to ecosystem health, aquaculture industries and wild-stock harvests, and physical infrastructures (WDFW 2011). The Bering Sea Marine Invasive Species Assessment: Ecosystem function and processes: did not expect *M. manhattensis* to impact ecosystem function in the Bering Sea (https://accs.uaa.alaska.edu/wp-content/uploads/molgulamanhattensis.pdf). However, the species has a high production rate of solids per unit tissue weight relative to other suspension feeders such as oysters and could influence biodeposition, transport, and composition of suspended sediments in estuaries (Haven and Morales-Alamo, 1966).

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Molgula manhattensis* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. There is moderate potential for impacts given the hard exterior tunic of this species and how often it is fouled.

Background Information: Molgula manhattensis may transport viable cells and cysts of toxic phytoplankton (Rosa et al., 2013). Dinoflagellates of the genus *Alexandrium* (formerly *Gonyaulax*), two species of which were transported during testing, have not been reported from the HB LME (Stewart and Howland, 2009; and references therein). Their establishment in the HB LME could have considerable and lasting negative impacts. Blooms of these toxic algae can cause red tides that negatively impact shellfish, fish, mammals, and birds that come into contact with the algae. Bivalve molluscs that consume these algae accumulate the toxins in their soft tissues, and when eaten can cause paralytic shellfish poisoning (PSP). The introduction of a

Japanese species of red tide dinoflagellate has had considerable impact on Australian shellfish industries (Hallegraeff, 1998). *Molgula manhattensis* may also host the commensal copepod *Lichomolgus albens* Thorell, 1859 (Rasmussen, 1973; cited in Jensen 2010) and may have a symbiotic relationship with a fungus-like protist, *Nephromyces* sp. which is likely associated with urate metabolism (Saffo and Davis, 1982, Saffo, 1988). In Beaufort, VA, the tunic of native *M. manhattensis* is often covered by hydroids, *Polydora*, *Corophium*, small *Balanus*, and *Membranipora* (Otsuka and Dauer, 1982). The likelihood of these taxa hitchhiking long distances with *M. manhattensis* on fouled vessels is unknown.

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Molgula manhattensis* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native tunicates of the same genus in the RA area, but no evidence of hybridization was found in the literature.

Background Information: No reports were found of *M. manhattensis* hybridizing with other species that occur in the HB LME. According to https://obis.org/ *M. griffithsii*, *M. retortiformis*, and *M. siphonalis* are present in the RA area, although it is unknown if they could hybridize.

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Molgula manhattensis* would have low or no impact on at-risk or depleted species in the RA area.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay; at-risk fish, bird, and mammal species are unlikely to be affected. There would be little overlap and there are few mechanisms to suggest risk since this species mainly impacts fouling organisms and there are no at-risk fouling species in the RA area.

Q16- Impact on fisheries (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Molgula manhattensis* would have low or no impact on fished species in the RA area. There is some doubt regarding possible cascading effects from fouling in the greater environment.

Background Information: Within its native range, *M. manhattensis* is a serious fouler of live oysters and oyster cultch, which consists of empty shells placed to encourage spat settlement (Andrews, 1953, Andrews, 1973; cited in Fofonoff et al., 2003). However, shellfish are not cultivated in the HB LME (Stewart and Lockhart, 2005).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Molgula manhattensis* is invasive somewhere else in the world. There is solid evidence of the species' establishment and impacts in invaded areas.

Background Information: Molgula manhattensis has been introduced to Japan, Australia, Argentina, and the Pacific coast of North America; and is introduced or possibly cryptogenic in Europe (Fofonoff et al., 2003, Haydar et al., 2011). On the Pacific coast of North America, its introduced range is from Newport Beach, California to Prince Rupert, BC (Lambert and Lambert, 1998, Clarke Murray et al., 2011). This ascidian is native to the Atlantic and Gulf coasts of North America. In Atlantic Canada it has been reported in the Scotian Shelf/Bay of Fundy area of NS (Emberley and Clark, 2011), PEI (Thomas, 1970), and Gulf of St. Lawrence (Brunel et al., 1998). The species taxonomy is somewhat confused, with many synonyms (Fofonoff et al., 2003). Recent genetic studies consider *M. manhattensis* and *M. tubifera* to be synonymous and genetically distinct from *M. socialis*, which is presumed to be native to the northeast Atlantic (Haydar et al., 2011). Reproductive plasticity, high fecundity, and short generation time may explain the success of this species as a colonizer (Jensen, 2010).

Styela clava Herdman, 1881

Phylum: Chordata

Class: Ascidiacea

Order: Stolidobranchia

Family: Styelidae

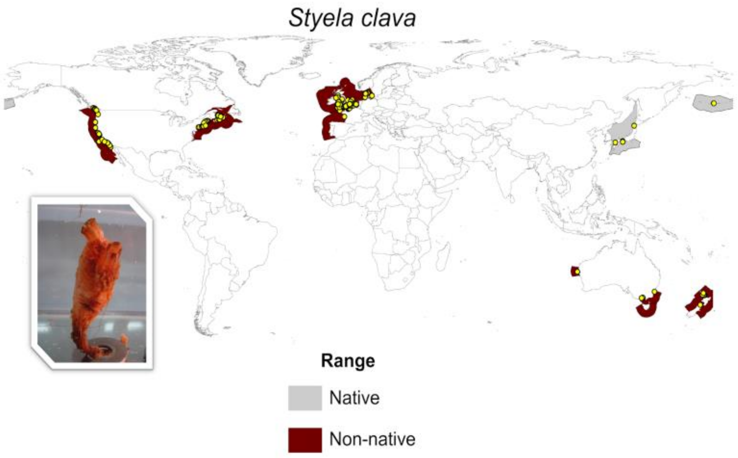

Figure 19: Ecoregions where Styela clava is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Styela clava occurrence points were obtained from OBIS (https://obis.org/). Picture of S. clava modified from https://invasions.si.edu/nemesis/species_summary/159337.

CMIST scores for *S. clava:*

Mean adjusted Likelihood of Invasion: 1.84

Mean adjusted Impact of Invasion: 1.87

Mean adjusted Overall CMIST score: 3.45

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Styela clava* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found of the species present in Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). *Styela clava* is established on the Atlantic (PEI, NS) (Vercaemer et al., 2012), and Pacific coasts of Canada (both sides of the Strait of Georgia) (Clarke and Therriault, 2007).

Q2- Rate of introduction **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Styela clava* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Styela clava is present in major European source ports such as Hamburg, Germany and Amsterdam, Netherlands (Chan et al., 2013). Propagule pressure is unknown but shipping may expose many Arctic ports, including Churchill, Deception Bay, Iqaluit, and Milne Inlet to this species through transport in ballast water, or on fouled hulls (Chan et al., 2012).

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Styela clava*. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the habitat classification.

Background Information: Styela clava is a low intertidal to subtidal fouling species that typically occurs at 15-25 m depth but is found to 40 m depth (Lützen, 1998, Therriault and Herborg, 2008b). It is most common in sheltered habitats with low wave action, such as inlets, bays, harbours and marinas (Lützen, 1998). It has also been found on high energy outer coasts on the partially protected surfaces of rocks or pilings, and can withstand tidal emersion in some microhabitats (e.g., under rocks). *Styela clava* attaches to a range of artificial substrates, including pier pilings, jetty walls, concrete structures, submerged ropes, buoys, floating docks, and vessel hulls (Lützen, 1998, NIMPIS, 2016). As a secondary settler it can settle on substrates already fouled by other species. *S. clava* is also found on natural substrates such as rocks and bivalve beds, and epiphytically on *Crassostrea gigas*, *Mytilus edulis*, and *Sargassum muticum* (Lützen, 1998, NIMPIS, 2016). In both its native and introduced range *S. clava* is more frequently reported from anthropogenic structures than from natural surfaces (Simkanin et al., 2012). More study is needed to determine whether this pattern is a sampling artifact and what is limiting establishment if it is not.

Q4- Survival (suitable environment) **(Score = 2, Certainty = 3)**

Score Rationale : Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Styela clava*. The species can survive under ice cover at sub-zero temperatures in other locations (e.g., Gulf of St. Lawrence), but these habitats are at the extreme northern extent of its distribution and winters would be shorter and ice cover thinner than in much of the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Styela clava can withstand temperatures ranging from ca. 0 to 23 °C (Buizer, 1980, Lützen, 1998; and references therein). It is present north to the Maritimes and Gulf of St Lawrence along the Atlantic coast (Vercaemer et al., 2012), where it overwinters under the ice in waters that reach sub-zero temperatures. Growth rate in Limfjord, Denmark decreases in November and growth is slow, if not completely arrested, during December through April at temperatures of 0-5 °C (Lützen, 1998). Davis and Davis (2007) did not find evidence of *S. clava* occurring in European waters where the summer temperature did not exceed 16 °C. Adult *S. clava* can withstand temporary drops in salinity to as low as 8‰, presumably by closing their siphons (Lützen, 1998). Distributional data suggest the species may have a long-term lower salinity tolerance of ca. 20‰ and upper salinity tolerance of <35‰

(Lützen, 1998, Davis and Davis, 2007). In the laboratory, adult mortality increased as salinity was reduced from 31‰ to 16‰. A salinity of 18‰ is deleterious to the larvae, which metamorphose in salinities between 20‰ and 32‰ (Kashenko, 1997). *Styela clava* is abundant in Massachusetts harbours with poor water quality (i.e., high levels of nitrogen, low oxygen (hypoxia), no eelgrass beds and poor water clarity) and fair as well as good water quality (Carman et al., 2007). Further evaluation is needed of the species' low temperature tolerances.

Q5- Establishment (reproductive requirements) (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are almost never available for *Styela clava* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with a maximum sea surface temperature of >15°C, the lowest observed limit for reproduction. Testing has not been conducted at temperatures below 15 °C, so it is uncertain if reproduction could be completed at lower temperatures.

Background Information: Styela clava is hermaphroditic but not self-fertile, as it has external fertilization (oviparous) (Holmes, 1969). The planktonic larvae do not feed but receive nutrients from the yolk sac (lecithotrophic). After hatching they remain planktonic for 24-48 hrs at 20 °C before settling, attaching, and metamorphosing on the substrate. Davis and Davis (2007) did not find *S. clava* in European waters that remained <16 °C in summer, and (Holmes, 1969) did not find gonad development at temperatures <16 °C. Eno et al. (1997) and other sources have suggested that *S. clava* is unable to reproduce at temperatures <15 °C but without providing supporting references or data, however Wong et al. (2011) observed spawning did not occur in New Zealand until water temperatures were >15 °C. Larvae are present in the Gulf of St. Lawrence from late June through late September and in New England between June and November with peaks between July and October (Clarke and Therriault, 2007). These ranges, when compared with temperature data (Chassé et al., 2014) suggest that spawning may occur at lower temperatures. Further review or study is needed to verify the species' temperature requirements for successful reproduction.

Styela clava are conspicuously absent in a turbid environment in Prince Edward Island (Orwell Bay), despite being extremely abundant in nearby, less turbid environments (McLaughlin et al., 2013). A thin layer of sediment deters larval settlement by *C. intestinalis*, and clogs the filtration apparatus of settlers (Lowen et al., 2016a), and may do the same for *S. clava*.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Styela clava*. Several studies show species-specific effects that could slow population growth. None of these species is known to occur in the RA area, but similar types of species, if present, might exert predation pressure.

Background Information: Styela clava along the western Atlantic coast are eaten by dove snails (*Astyris lunata*, *Anachis lafresnayi*, *Anachis avara*), fish (cunner *Tautogolabrus adspersus*) (Osman and Whitlatch, 2000, Locke et al., 2007; and references therein), and rock crab (*Cancer irroratus*) (Carver et al., 2006b). No reports were found of these species occurring in the HB LME. Predation of *A. lunata* and perhaps other species by cunner and green crab (*Carcinus maenas*) may facilitate the establishment and growth of *S. clava* populations (Locke et al., 2007).

From Clarke and Therriault (2007): "*Styela clava* experiences ontogenetic and size-selective shifts in predation pressure. Larvae are susceptible to predation by a range of planktivorous predators. Newly settled ascidian juveniles become prey for mobile carnivorous or omnivorous invertebrates and fish (reviewed in Lambert, 2005). For example, in New England, juveniles are susceptible to high mortality (almost 100%) from snails such as *Mitrella lunata* and fish such as *Tautogolabrus adspersus* (Osman and Whitlatch, 2000, NIMPIS, 2002). The presence of these predators seemed to prevent the establishment of *S. clava* populations in these areas (Osman and Whitlatch, 1995). Adult *Styela clava* have a tough, leathery tunic and no predators have been recorded for north-western Europe (Lützen, 1998, Cohen, 2005, NIMPIS, 2002). Predators recorded for other adult solitary ascidians include flatworms, gastropods, sea stars, crabs, fish and birds (Lambert, 2005). Many ascidians deter predation and fouling by producing noxious secondary metabolites. However, *S. clava* may employ a different strategy: the thickening, tubercles and folds of the tunic promote fouling by other organisms such that individuals become cryptic (Lambert, 2005). The rapid growth rate allows them to escape predation through size at an earlier age than comparable native species (Osman and Whitlatch, 1995). Predation on small recruits and juveniles can completely eliminate this species from some areas, even though transplanted adults may survive there (Osman and Whitlatch, 1995)."

Q7- Spread (potential natural dispersal) (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Styela clava* has the capacity for a very limited range of natural dispersal in the RA area. There is speculation that natural dispersal may occur when sessile adults are attached to objects, but good knowledge of the species' short larval duration, which limits dispersal at the larval life stage.

Background Information: In Malpeque Bay, PEI, the prevailing winds and water flow patterns during the reproductive phase may have influenced local dispersal of *S. clava* (Arsenault et al., 2009). In this area the planktonic larvae are only viable for ca. 24 hours in the water column (see Q.5), so their natural dispersal may be limited to a few km annually. Most range extensions likely occur after settling when sessile adults hitchhike on natural or artificial substrates. In the HB LME the larval period will remain short even if colder water delays maturation, with dispersal occurring primarily by hull fouling.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good evidence for wide range dispersal through biofouling which could occur on vessels visiting the RA area, as well as on fishing and recreational vessels that move within the region.

Background Information: Hull fouling on domestic coastal resupply and other boat traffic could spread *S. clava* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Clarke Murray et al., 2011, Chan et al., 2012). Its strong attachment and low drag coefficient enable *S. clava* to avoid dislodgement at flow velocities well above those of fast moving vessels (i.e., >70 km/h) (Clarke Murray et al., 2012). Transport of larvae between communities in ballast water is also a possibility (Clarke and Therriault, 2007).

There is no aquaculture in the region at present to attract *S. clava* that could be transported with fouled equipment. With the exception of ship-related movements, the importance of other types of movements (e.g., smaller pleasure and fishing vessels) in HBC is not well known.

Q9- Impact on population (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. The species is known to impact many populations in invaded areas although these are largely limited to artificial structures, so severe effects would likely be limited to a few areas in the RA area.

Background Information: Even medium-sized *S. clava* are taller than most other sessile organisms on a surface, which enables the species to compete effectively as a filtrator (Lützen, 1998). Intense filtration of plankton from the water column by *S. clava* can reduce the food available to co-occurring species and thereby their growth rate; reduce settlement of other fouling organisms through predation on their planktonic larvae (Osman and Whitlatch, 2000); and may lower the carrying capacity of bays in which blue mussels (*Mytilus edulis*) are cultured (Comeau et al., 2015). At sites in Massachusetts and New Hampshire *S. clava* increased slowly in abundance but without evidence of competitive dominance, perhaps due to limiting by physical conditions (Berman et al., 1992). In areas south of New Hampshire where it is abundant the species has achieved competitive dominance (Berman et al., 1992). In English waters the introduction and increase of *S. clava* was matched by a concurrent decline in the population of the local ascidian, *Ciona intestinalis* (Lützen, 1998), and in southern California *S. clava* has effectively replaced the indigenous ascidians (Lambert and Lambert, 1998).

Q10- Impact on communities (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* would have high impact in few areas or moderate impact in many areas on communities in the RA area. The species is known to impact many populations in invaded areas, although these are largely limited to artificial structures, so any severe effects should be localized to a few areas in the RA area.

Background Information: Styela clava introduced to shallow, protected habitats can replace native ascidians, becoming the dominant ascidian species and altering community biodiversity and species richness (Lambert and Lambert, 1998, 2003). In southern California harbours, *S. clava* and other introduced ascidians have replaced the native species *Pyura haustor* and *Ascidia ceratodes* (Lambert and Lambert, 1998). While *S. clava* has the ability to reduce settlement rates of co-occurring species locally it appears to have little effect on post-settlement community dynamics (Whitlatch et al., 1995; cited in Therriault and Herborg, 2008a).

In experimental studies with various densities of *Styela clava*, fauna from Port Phillip Bay, Australia showed idiosyncratic responses to increased *Styela* density, with crustaceans, and tanaids, as a group decreasing, while the deposit-feeding bivalve *Laternula rostrata* increased. These changes appeared to be due to subtle changes in hydrodynamics, sediment quality, emigration or altered survival in *Styela* rather than community-wide effects (Ross et al., 2007).

Q11- Impact on habitats (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. The species is known to impact pelagic and benthic habitats in invaded areas although these are largely limited to artificial structures, so any severe effects should be localized to few areas in the RA area.

Background Information: The dense aggregations formed by *Styela clava* can alter habitat structure, excluding some organisms and providing secondary substrate for other fouling organisms (e.g., ascidians) (Lambert and Lambert, 1998, 2003; Lützen, 1998). The tunics of *S. clava*, which is a relatively large species, are often fouled by smaller colonial tunicates (Therriault and Herborg, 2008b). In high densities, *C. intestinalis* and *Styela clava* can reduce plankton loads in enclosed embayments used for aquaculture (Comeau et al., 2015), and substantially increase biodeposition of fecal pellets and ascidian carcasses to the bottom sediment (McKindsey et al., 2009, Qi et al., 2015). However, there are no shellfish aquaculture operations in the HB LME.

Q12- Impact on ecosystem function (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* would have high impact in a few areas or moderate impact in many areas on ecosystem function in the RA area. The species is known to impact pelagic and benthic habitats in invaded areas although these are largely limited to artificial structures, so any severe effects should be localized to few areas in the RA area.

Background Information: Styela clava has replaced local ascidians as the dominant member of shallow, protected habitats worldwide. In high densities, *C. intestinalis* and *Styela clava* can reduce plankton loads in enclosed embayments used for aquaculture (Comeau et al., 2015), and substantially increase biodeposition of fecal pellets and ascidian carcasses to the bottom sediment (McKindsey et al., 2009, Qi et al., 2015). In laboratory experiments *Styela clava* was a very efficient predator of oyster (*Crassostrea virginica*) larvae in the water column (Osman et al. 1989). However, there are no shellfish aquaculture operations in the HB LME.

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Styela clava* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. There is moderate potential for impacts given the hard exterior tunic of this species and how often it is fouled.

Background Information: Little is known about the parasites and pathogens of *S. clava* but effects from fellow travelers could exist. Larger specimens of *S. clava* are often covered with epibionts such as corals, algae, hydroids, sponges, or other tunicates (Therriault and Herborg, 2008b). The latter can include the non-indigenous colonial tunicates, *Botryllus schlosseri* and *Botrylloides violaceus*, which could be spread with *S. clava*. In Argentina, colonization on the holdfast of *S. clava* by sporelings may have facilitated invasion by the seaweed *Undaria pinnatifida* (Pereyra et al., 2015).*Styela clava* and other ascidians may also transport viable cells and cysts of toxic phytoplankton (Rosa et al., 2013).

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Styela clava* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native tunicates of the same genus in the RA area, but no evidence of hybridization was found in the literature.

Background Information: Styela coriacea (Alder and Hancock, 1848) is present in the HB LME (Stewart and Lockhart, 2005) but no information was found on possible hybridization with *S. clava*. The potential genetic impacts of epibionts and other travelers are uncertain.

Q15- Impact on at-risk species (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* would have low or no impact on at-risk or depleted species in the RA area. There is little overlap or mechanisms to suggest risk since it is mainly associated with impacts on artificial structures.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay; at-risk fish, bird, and mammal species are unlikely to be affected. There would be little overlap and there are few mechanisms to suggest risk since this species mainly impacts fouling organisms and there are no at-risk fouling species in the RA area.

Q16- Impact on fisheries (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* would have low or no impact on fished species in the RA area. There would be little overlap and there are few mechanisms to suggest risk since this species impacts primarily farmed organisms in suspension and there are no shellfish aquaculture operations in the RA area.

Background Information: Styela clava is a major pest of oyster and mussel farms in eastern Canada (Clarke and Therriault, 2007, Locke et al., 2007, Arsenault et al., 2009). Shellfish are not cultivated in the HB LME. Natural populations of blue mussel (*Mytilus edulis*) that have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, and occasionally for subsistence by Hudson Bay residents (Stewart and Lockhart, 2005) are bottom-growing and unlikely to be impacted. Damage to mussel populations in the Hudson Bay LME that are important prey for Hudson Bay eider (*Somateria mollissima sedentaria*) that overwinter in polynyas in the Belcher Islands, where they eat mussels and are hunted for subsistence, is also unlikely.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Styela clava* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Styela clava is native to the Northwest Pacific, from Shanghai to the Sea of Okhotsk and the southeastern Bering Sea (Lützen, 1998, Fofonoff et al., 2003; and references therein). Its strong attachment and low drag coefficient enable *S. clava* to avoid dislodgement at flow velocities well above those of fast moving vessels (i.e., > 70 km/h) (Clarke Murray et al., 2012). The species has become widely distributed in coastal waters, through fouling of ships and recreational craft and transport with oysters (Fofonoff et al., 2003, Clarke and Therriault, 2007; and references therein; Clarke Murray et al., 2011). It has been introduced to both coasts of North America (NA) and to Europe, Australia and New Zealand. On the Atlantic coast of NA it is distributed from Virginia north to Nova Scotia (Moore et al., 2014) and Prince Edward Island (Locke et al., 2007, Ramsay et al., 2008, Arsenault et al., 2009). On the Pacific coast of NA it is distributed from California to Vancouver Island, British Columbia (Lambert, 2003). In NS the establishment of *S. clava* may be facilitated by the provision of artificial structures, particularly related to aquaculture, since natural hard substrates are scarce (Locke et al., 2007). The green crab (*Carcinus maenas*) may also facilitate the spread of *S. clava* by eating the snail *Astyris lunata* which preys upon *S. clava*.

ZOOBENTHOS – OTHER

***Cordylophora caspia* (Pallas, 1771)**

Phylum: Cnidaria
  Class: Hydrozoa
    Order: Anthoathecata
      Family: Cordylophoridae

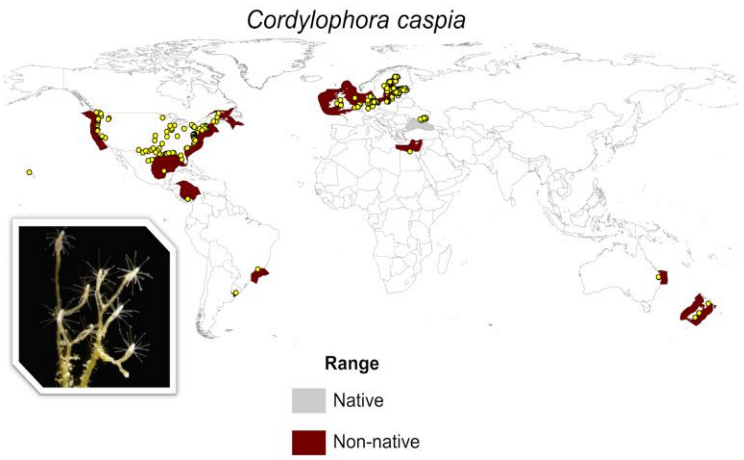

Figure 20: Ecoregions where Cordylophora caspia is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Cordylophora caspia occurrence points were obtained from OBIS (https://obis.org/) and NEMESIS (http://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893). Note: this species can also be present in freshwater ecosystems, so inland points are included on the map. Picture of C. caspia modified from https://invasions.si.edu/nemesis/species_summary/48893.

CMIST scores for *C. caspia:*

Mean adjusted Likelihood of Invasion: 1.95

Mean adjusted Impact of Invasion: 1.89

Mean adjusted Overall CMIST score: 3.69

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found of the species being present in Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Goldsmit, 2016).

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Cordylophora caspia is present in ports that are directly connected by shipping to Churchill, Deception Bay, and Iqaluit and could potentially be entrained and transport by these vessels in ballast water, or on fouled hulls (Folino, 2000, Bij de Vaate et al., 2002, Pienimäki and Leppäkoski, 2004, Janssen et al., 2005, Streftaris et al., 2005, Cohen, 2006, Fofonoff et al., 2009, Chan et al., 2012).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Cordylophora caspia*. None of the species known to act as substrates for *C. caspia* are present in the assessment area, but other physical structures could serve as settlement surfaces. There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the habitat classification.

Background Information: Cordylophora caspia is a sessile hydrozoan that lacks a planktonic medusa stage. There may be multiple species of *Cordylophora* with different habitat preferences (Folino-Rorem et al., 2009).

From NEMESIS: "It occurs in estuaries, lagoons, and coastal lakes (Arndt, 1989), rarely in oligohaline freshwater (Bij de Vaate et al., 2002). Planulae of *C. caspia* settle and grow on a wide range of substrates, including shells, rock, wood, and vegetation. This hydroid has been found on a number of plants, including submerged plants (*Ceratophyllum demersum*- Coontail; *Nitella* sp.; *Potamogeton* sp.- Pondweeds; *Elodea* sp.- Waterweed; *Vallisneria americana*- Wild Celery), stalks of floating plants (*Nymphaea odorata*- White Water Lily), and roots and stems of emergents (*Alternanthera philoxeroides*- Alligatorweed; *Phragmites australis* - Common Reed). It has also been reported from shells of living freshwater (Zebra and Quagga Mussels) and brackish-water mussels (*Mytilopsis leucophaeata*). It has also been found on man-made substrates including old automobiles and nylon ropes, buoys and ships" (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893 and references therein; Bij de Vaate et al., 2002)). Zebra mussels (*Dreissena polymorpha*) are invading the Nelson River system which drains into Hudson Bay (https://www.hydro.mb.ca/environment/wildlife_stewardship/zebra_mussels/), and when they reach the estuary might be a dispersal vector providing substrate for *C. caspia.*

Q4- Survival (suitable environment) (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest the RA area offers a negligible proportion of suitable environmental conditions for *Cordylophora caspia*. It is tolerant to different salinities depending on temperature. The species is known to be more prevalent in freshwater systems. The limited information available on thermal tolerances suggests the genus only survives above 2°C. Published tolerances are for the genus, generally, and not a lot is known about the species, however distribution information suggests it may be able to survive at temperatures down to zero in freshwater.

Background Information: From CABI: "In general, it is difficult to propose an optimal salinity since various temperatures have been used in laboratory studies and endogenic differences exist for an organism with ecologically plastic physiological responses to salinity and temperature (Arndt, 1984)." (https://www.cabi.org/isc/datasheet/113452)

From Folino-Rorem et al. (2009): "The published salinity range for *Cordylophora* is 0–40 PSU, though 15–17 PSU has been cited as the optimal range for brackish populations (Kinne, 1958, Arndt, 1984)".

From CABI: "Colonies of *C. caspia* develop well at 2–12‰ salinity with relatively constant environmental conditions, and at 2–6‰ with considerable tidal influence (Arndt, 1989). Tolerated temperature is 2-24 °C, with range influenced by salinity (Kinne, 1971)" (https://www.cabi.org/isc/datasheet/113452).

From Bij de Vaate et al. (2002): "The occurrence of *C. caspia* has also been reported from upper reaches of rivers at favourable conditions such as fast flow, high oxygen availability, and positive ion anomalies (Ca^{2+}, Mg^{2+}, Na^{+}) (Arndt, 1989)."

From NEMESIS: “In more severe climates, such as the Netherlands and Finland, hydroids may be dormant in winter (Roos, 1979, Jormalainen et al., 1994) and based on distribution in freshwater can survive as low as 0 °C” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are sometimes available for *Cordylophora caspia* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with minimum sea surface salinity <= 27 ppt and maximum sea surface temperature >= 5 °C, including most Hudson Bay coast and all of James Bay). Good experimental data are available on the species’ reproductive requirements for comparison with environmental layers to evaluate the extent of conditions suitable for *C. caspia* reproduction in the RA area.

Background Information: From Bij de Vaate et al. (2002): “The species has no medusal stage, its sexual stage is at the polyp, and planula larvae released settle after a short planktonic life. Asexual dispersal mechanisms, e.g., body fragments (hydrorhiza) and menonts (fragment of tissue with potential regenerative capabilities), which survive in wintertime, contribute to its colonization success in Europe (Bij de Vaate et al., 2002). Menonts survive in sea water and can even develop into polyps at maximum salinity of 40‰ (minimum: 0.08‰) (Vervoort, 1946).”

Colonial hydroids can have continual cellular regeneration. As a consequence, the lifetime reproductive output of the colonies approaches infinity, and because of partial mortality, fragmentation, and fusion, colony size correlates poorly with age (Jormalainen et al., 1994; and references therein). In both freshwater and brackish populations in temperate locations, colonies begin to come out of dormancy and regenerate hydranths in the spring when temperature is between 6 and 15 °C (Musko et al., 2008).

Colony growth increases with temperatures at >5 °C and gonophores are produced when temperatures reach 10 °C in brackish (Arndt, 1989) or around 15-18 °C in freshwater (Musko et al., 2008; and other references; https://www.cabi.org/isc/datasheet/113452).

Upper salinity ranges are based on experimental survival. The upper limit for sexual reproduction was 27 ppt (Kinne, 1958) (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents could slow population growth in the RA area for *Cordylophora caspia.* There is substantial information on predation of *C. caspia* in other regions and closely related species are present in the RA, but there is no evidence that they would severely limit population growth.

Background Information: Cordylophora caspia is preyed upon by *Tenellia adspersa*; amphipods (*Gammarus* spp.) and fish (*Tridentiger bifasciatus*) (NEMESIS and CABI). *Gammarus tigrinus* has not been reported from the RA area, but other species of the genus are present (e.g., *G. oceanicus*, *G. setposus*, *G. wilkitzkii*; Stewart and Lockhart, 2005). The degree of natural control they might exert on introduced *C. caspia* is unknown.

From NEMESIS: "*Cordylophora caspia* is an important food for nudibranchs, which include many specialized predators of hydroids. *Cordylophora caspia* is apparently eaten by the nudibranch *Tenellia adspersa*, cryptogenic on the East Coast of North America, but widely introduced elsewhere (Gaulin et al., 1986, Chester, 1996). Despite its nematocysts, *C. caspia* is also eaten by some generalized predators, such as amphipods (Roos, 1979). Extensive feeding by the introduced amphipod *Gammarus tigrinus* (native in Chesapeake Bay) on *C. caspia* was reported in Dutch freshwaters by Roos (1979). In the San Francisco Bay estuary, *C. caspia* comprised 18-23% of the diet of the introduced Shimofuri goby (*Tridentiger bifasciatus*) (Matern and Brown, 2005)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893)

From CABI: "A well-known predator on brackish *Cordylophora* colonies is the estuarine nudibranch, *Tenellia adspersa*. *Tenellia* is a generalist nudibranch but is often feeds on brackish colonies of *Cordylophora* causing a noticeable decline in colonies during late summer and early autumn in temperate areas (Harris and Lg, 1980, Gaulin et al., 1986, Arndt, 1989, Chester, 1996, Blezard, 1999). Furthermore, predation by *Tenellia adspersa* (a synonym of *Embletonia pallida*) may graze so heavily on *Cordylophora* (*caspia*) colonies causing predator-induced dormancy (Jormalainen et al., 1994, Jewett, 2005). Blezard (1999) demonstrated that fecundity and development of *Tenellia* were less than optimal at salinities below 12 PSU creating a salinity refuge from predation for *Cordylophora* (*lacustris*). Another invertebrate predator of *Cordylophora* hydranths is the amphipod, *Gamarus* (Roos, 1979)." https://www.cabi.org/isc/datasheet/108329.

Q7- Spread (potential natural dispersal) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* has the capacity for a moderate range of natural dispersal in the RA area. There is moderate evidence available to suggest a reasonable dispersal range via drifting by attached objects, however the short larval duration limits dispersal at this stage.

Background Information: Planula larvae released by *C. caspia* are only mobile for a short period, swimming or crawling for less than 24 h before settling, which limits larval dispersal away from the parent colony (Sommer, 1992, Gili and Hughes,1995) and via ballast water. Asexual dispersal mechanisms, e.g., body fragments (hydrorhiza) and menonts (fragment of tissue with potential regenerative capabilities), which survive in wintertime, contribute to its colonization success in Europe (Bij de Vaate et al., 2002).

From CABI: "In its menont stage *C. caspia* can easily be transported over large distances, because in this stage the species is temperature and drought resistant (Bij de Vaate et al., 2002). Dispersal could occur via floating plant material drift (Roos, 1979, Koetsier and Bryan, 1989), and perhaps via birds (Davis, 1957, Green and Figuerola, 2005, Musko et al., 2008)." (https://www.cabi.org/isc/datasheet/108329)

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Cordylophora caspia* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is much solid evidence for wide range dispersal through uptake in ballast and biofouling which could occur through vessels moving within the RA area.

Background Information: Invasion pathways include the aquarium trade (Mills et al., 1993), and shipping via ballast water and/or hull fouling (Ruiz et al., 1997, Seyer et al., 2017). *Cordylophora* populations in the Great Lakes spread predominantly through sexually produced propagules, and while limited natural larval dispersal can drive population expansion locally, regional expansion likely relies on anthropogenic dispersal vectors (Darling and Folino-Rorem, 2009). Chan et al. (2012) suggest that there is considerable domestic "Arctic direct" ballast exchange (about half of total) in the HB LME, suggesting that this is a plausible vector for within-region spread. Biofouling and ballast transported by domestic coastal resupply and other boat traffic, could spread *C. caspia* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012, Chan et al., 2015).

From Bij de Vaate et al. (2002): "*C. caspia* has been dispersed via canals and rivers attached to ships' hulls, floating reed, water lily stems, and other macrophytes (Roos, 1979), and in ballast water for overseas transportation. In its menont stage *C. caspia* can easily be transported over large distances, because in this stage the species is temperature and drought resistant.

From CABI: "The global spread and establishment of *C. caspia* is primarily attributed to increased ship transport through canals and rivers via ship ballast and/or hull fouling (Folino, 2000, Bij de Vaate et al., 2002, Pienimäki and Leppäkoski, 2004, Janssen et al., 2005, Streftaris et al., 2005). In addition, the likelihood of new introductions continues with increased shipping (spread via ship ballast and/or hull fouling)." (https://www.cabi.org/isc/datasheet/108329)

Q9- Impact on population (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. There are known effects on various populations, although most are indirect as a result of habitat alteration.

Background Information: From CABI: "*Cordylophora* competes with other fouling organisms such as mussels and hydroids (Smit et al., 1993, Jewett, 2005). *Cordylophora* competes with bryozoan (*Victorella pavida*) and ciliates for substrate space (Jewett, 2005).

The potential impact on habitats by *Cordylophora* is extensive since the organism inhabits freshwater and brackish aquatic habitats of various types (Zaiko et al., 2007). *Cordylophora* can become very abundant and modify habitats by growing on submerged substrata on soft bottoms changing the community structure of soft bottoms (Olenin and Leppäkoski, 1999). Colonies are capable of creating refuges for and from predators and currents and also assist in the accumulation of particulate organic matter (Leppäkoski, 2004). The filamentous structure of *Cordylophora* colonies may also serve to enhance the settlement and recruitment of dreissenid mussel larvae (Folino-Rorem and Stoeckel, 2006) and the establishment of macroinvertebrates in zebra mussel colonies by providing more surface area (Moreteau and Khalanski, 1994, Folino-Rorem et al., 2006).

In freshwater systems, zebra mussel recruitment is highest in areas of increased vegetation (Stańczykowska and Lewandowski, 1993), though this depends on plant architecture (Cheruvelil et al., 2002, Kraufvelin and Salovius, 2004). In addition macrophytes also provide macroinvertebrates with refugia from fish predation (Dykman and Hann, 1996, Warfe and Barmuta, 2004, Harrison, 2005). Epiphytes, such as the filamentous alga *Cladophora* or filamentous epifauna such as the hydroid *Cordylophora*, may enhance macroinvertebrate colonization of mussel colonies. Folino-Rorem et al. (2006) attributed increased zebra mussel settlement on artificial filamentous substrata (hydroid mimics) to an increase in total surface area rather than a preference for filamentous substrata suggesting that settlement of zebra mussel larvae and other macroinvertebrates on aquatic macrophytes may simply be a function of the increase in substrate surface area afforded by these filamentous organisms. These same filaments facilitated the colonization of chironomids and caddisflies; chironomid and caddisfly densities were significantly greater than on control or plates with no filaments (chironomids: $p < 0.003$; caddisflies: $p < 0.008$) (J Stoeckel & N Folino-Rorem, Wheaton College, Illinois, USA, personal communication, 2009)." (https://www.cabi.org/isc/datasheet/108329)

From NEMESIS: "*Cordylophora caspia* is a potential competitor for space in fouling communities. In field experiments on fouling plates (Key Bridge, Patapsco River, Maryland), where laboratory-grown colonies of *C. caspia* were added, abundances of the bryozoan *Victorella pavida* (cryptogenic), the entoproct *Loxosomatoides laevis* (introduced), and the protozoans *Metafolliculina* sp., and *Stentor* sp. were reduced (Von Holle and Ruiz, 1997; Von Holle unpublished data).

Although colonies of *Cordylophora caspia* in many bodies of water represent a substantial biomass of predators on zooplankton and mobile epibenthos (Bibbins, 1892, Roos, 1979, Arndt, 1984), their role as predators has rarely been studied quantitatively. However, *C. caspia* predates on settling Zebra Mussel (*Dreissena polymorpha*) veligers, selecting smaller veligers, even as their filaments increase overall rates of settlement (Folino-Rorem and Stoeckel, 2006)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893)

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There are known effects on various communities, although most are indirect as a result of habitat alteration.

Background Information: From CABI: "The potential impact on habitats by *Cordylophora* is extensive since the organism inhabits freshwater and brackish aquatic habitats of various types (Zaiko et al., 2007). *Cordylophora* can become very abundant and modify habitats by growing on submerged substrata on soft bottoms changing the community structure of soft bottoms (Olenin and Leppäkoski, 1999). Colonies are capable of creating refuges for and from predators and currents and also assist in the accumulation of particulate organic matter (Leppäkoski, 2004). The filamentous structure of *Cordylophora* colonies may also serve to enhance the settlement and recruitment of dreissenid mussel larvae (Folino-Rorem and Stoeckel, 2006) and the establishment of macroinvertebrates in zebra mussel colonies by providing more surface area (Moreteau and Khalanski, 1994, Folino-Rorem et al., 2006).

In freshwater systems, zebra mussel recruitment is highest in areas of increased vegetation (Stańczykowska and Lewandowski, 1993), though this depends on plant architecture (Cheruvelil et al., 2002, Kraufvelin and Salovius, 2004). In addition macrophytes also provide macroinvertebrates with refugia from fish predation (Dykman and Hann, 1996, Warfe and Barmuta, 2004, Harrison, 2005). Epiphytes, such as the filamentous alga *Cladophora* or filamentous epifauna such as the hydroid *Cordylophora*, may enhance macroinvertebrate colonization of mussel colonies. Folino-Rorem et al. (2006) attributed increased zebra mussel settlement on artificial filamentous substrata (hydroid mimics) to an increase in total surface area rather than a preference for filamentous substrata suggesting that settlement of zebra mussel larvae and other macroinvertebrates on aquatic macrophytes may simply be a function of the increase in substrate surface area afforded by these filamentous organisms. These same filaments facilitated the colonization of chironomids and caddisflies; chironomid and caddisfly densities were significantly greater than on control or plates with no filaments (chironomids: $p < 0.003$; caddisflies: $p < 0.008$) (J Stoeckel & N Folino-Rorem, Wheaton College, Illinois, USA, personal communication, 2009)." (https://www.cabi.org/isc/datasheet/113452)

From NEMESIS: "*Cordylophora caspia* is a potential competitor for space in fouling communities. In field experiments on fouling plates (Key Bridge, Patapsco River, Maryland), where laboratory-grown colonies of *C. caspia* were added, abundances of the bryozoan *Victorella pavida* (cryptogenic), the entoproct *Loxosomatoides laevis* (introduced), and the protozoans *Metafolliculina* sp., and *Stentor* sp. were reduced (Von Holle and Ruiz, 1997; Von Holle unpublished data).

Although colonies of *Cordylophora caspia* in many bodies of water represent a substantial biomass of predators on zooplankton and mobile epibenthos (Bibbins, 1892, Roos, 1979, Arndt, 1984), their role as predators has rarely been studied quantitatively. However, *C. caspia* predates on settling Zebra Mussel (*Dreissena polymorpha*) veligers, selecting smaller veligers, even as their filaments increase overall rates of settlement (Folino-Rorem and Stoeckel, 2006)." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893)

Q11- Impact on habitats (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. There are some reported effects on habitats related to this species.

Background Information: From CABI: "*Cordylophora* can become very abundant and modify habitats by growing on submerged substrata on soft bottoms changing the community structure of soft bottoms (Olenin and Leppäkoski, 1999). Colonies are capable of creating refuges for and from predators and currents and also assist in the accumulation of particulate organic matter (Leppäkoski, 2004). The filamentous structure of colonies may also serve to enhance the settlement and recruitment of dreissenid mussel larvae (Folino-Rorem and Stoeckel, 2006) and the establishment of macroinvertebrates in zebra mussel colonies by providing more surface area (Moreteau and Khalanski, 1994, Folino-Rorem et al., 2006). *Cordylophora* acts as a biofouler by colonizing power station cooling systems and fouling industrial water pipes in Europe and North America" (https://www.cabi.org/isc/datasheet/108329).

In freshwater systems, zebra mussel recruitment is highest in areas of increased vegetation (Stańczykowska and Lewandowski, 1993), though this depends on plant architecture (Cheruvelil et al., 2002, Kraufvelin and Salovius, 2004). In addition macrophytes also provide macroinvertebrates with refugia from fish predation (Dykman and Hann, 1996, Warfe and Barmuta, 2004, Harrison, 2005). Epiphytes, such as the filamentous alga *Cladophora* or filamentous epifauna such as the hydroid *Cordylophora*, may enhance macroinvertebrate colonization of mussel colonies. Folino-Rorem et al. (2006) attributed increased zebra mussel settlement on artificial filamentous substrata (hydroid mimics) to an increase in total surface area rather than a preference for filamentous substrata suggesting that settlement of zebra mussel larvae and other macroinvertebrates on aquatic macrophytes may simply be a function of the increase in substrate surface area afforded by these filamentous organisms. These same filaments facilitated the colonization of chironomids and caddisflies; chironomid and caddisfly densities were significantly greater than on control or plates with no filaments (chironomids: $p < 0.003$; caddisflies: $p < 0.008$) (J Stoeckel & N Folino-Rorem, Wheaton College, Illinois, USA, personal communication, 2009)." (https://www.cabi.org/isc/datasheet/108329)

Q12- Impact on ecosystem function (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Cordylophora caspia* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. There are known effects on various communities/species, although most are indirect as a result of habitat alteration. Effects on ecosystems have been reported, but mechanisms not demonstrated.

Background Information: The ecological impact of *Cordylophora* needs further exploration, though as a sessile raptorial suspension feeder, this predatory hydroid likely modifies aquatic trophic structures by feeding on larval fish prey (Olenin and Leppäkoski, 1999). *Cordylophora* feeds on chironomids, an important fish food (Menzie, 1981). This species can introduce new ecosystem functions (sessile raptorial suspension feeding) such as in the Baltic Sea (Olenin and Leppäkoski, 1999).

Several species may become quite abundant because of alterations in food availability (i.e. the species itself) and its habitat changes, protecting predators/prey from harsh conditions. (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893)

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Cordylophora caspia* would have low or no impact associated with diseases, parasites or travelers in the RA area. There is little information available and no evidence of impacts from disease/parasites.

Background Information: From Norwegian Biodiversity Information Centre: “The filamentous structure of *Cordylophora* provides a substrate to inhabit for macroinvertebrates such as chironomids and caddis flies (Roque et al., 2004, Folino-Rorem et al., 2006). Freshwater populations of *Cordylophora* are often associated with bryozoa (e.g. *Plumatella*), sponges and protozoa that colonize the stalks or perisarc of the colonies” (https://www.biodiversity.no and references therein).

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Cordylophora caspia* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: There are no *Cordylophora* species in the assessment region (https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Cordylophora caspia* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be impacted given the demonstrated effects on habitats and ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through food web and ecosystem impacts - these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and thorny skate, *Amblyraja radiata*. Walrus populations of risk/special concern may also be impacted as they feed on benthic invertebrates so may be impacted through food web changes in benthic habitats.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Cordylophora caspia* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. There is evidence of impacts through fish predation and of irritation of fish gills, however this is mainly based on effects of closely related hydroids on other salmonid species.

*Background Information: Cordylophora caspia is a p*redatory hydroid that can feed on larval fish (Olenin and Leppäkoski, 1999). Whether it would cause mortality among anadromous Arctic char (*Salvelinus alpinus*) or brook trout (*S. fontinalis*) is unknown. Both species are harvested for subsistence and char also for commercial sale (Stewart and Lockhart, 2005). From Bloecher et al. (2018) (information on another hydroid and on farmed salmon): “Contact with the hydroids’ (*Ectopleura larynx*) nematocysts has the potential to cause irritation and pathological damage to salmon gills. Amoebic gill disease (AGD), caused by the amoeba *Paramoeba perurans*, is an increasingly international health challenge in Atlantic salmon farming.”

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Cordylophora caspia* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: From NEMESIS: “*Cordylophora caspia* was first described from the Caspian Sea by Pallas in 1771 and is believed to be native to the Black Sea-Caspian Sea

region (Briggs, 1931, Naumov, 1969, Hutchinson, 1993). Shipping has spread *C. caspia* through much of the world and this hydroid is now known from temperate and tropical coastal regions of every continent (except Antarctica), and from many fresh waters as well (Naumov, 1969, Arndt, 1984, Slobodkin and Bossert, 1991, Hutchinson, 1993)."
(https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=48893)

From Bij de Vaate et al. (2002): "In 1924 the species already had a worldwide distribution, from the cold boreal and antiboreal to the subtropical regions (Roch, 1924, Arndt, 1989)."

From Folino-Rorem et al. (2009) and references therein: "Repeated introductions may contribute to the current global distribution of *Cordylophora*. The first records of *Cordylophora* in the Baltic Sea are from the early 1800s, and it was observed in Danish waters in 1895. It continued to spread throughout Western Europe to the Loire estuary, France and to inland German waters around 1858. The hydroid was first found on the east coast of North America in Mystic Pond, Massachusetts in 1860 and was later discovered on the West coast of North America in the Puget Sound and San Francisco Bay areas circa 1920. *Cordylophora* spp. supposedly invaded the Great Lakes via the St. Lawrence River System in 1956. In freshwater systems, *Cordylophora* is becoming a prevalent biofouler, possibly due to changes in water quality (increased salts) and its ability to colonize various hard substrata including zebra mussels."

Marenzelleria viridis **(Verrill, 1873)**

Phylum: Annelida
  Class: Polychaeta
    Order: Spionida
      Family: Spionidae

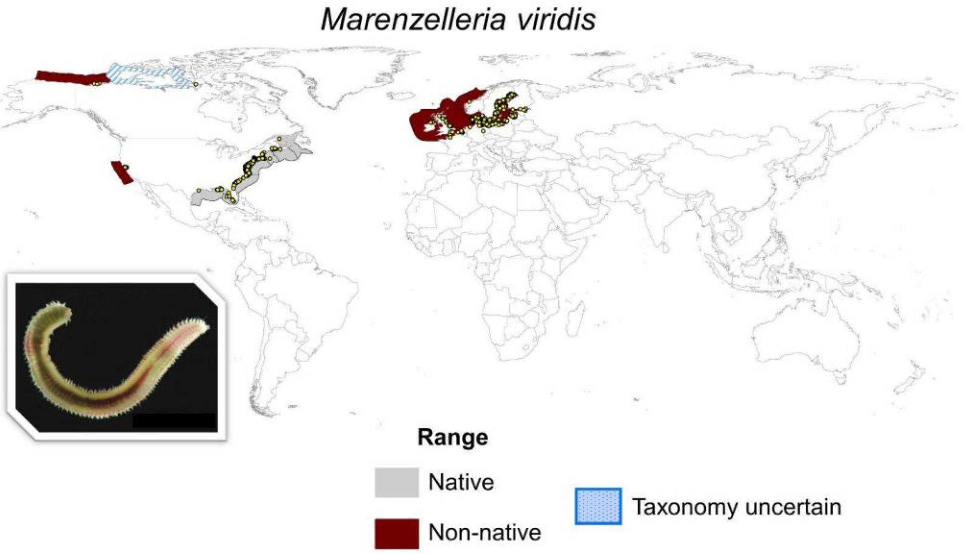

Figure 21: Ecoregions where Marenzelleria viridis is distributed: native (grey) and non-native (dark red) regions. Ecoregion where the status is taxonomically uncertain has been marked in blue hatching. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Marenzelleria viridis occurrence points were obtained from OBIS (https://obis.org/) and GBIF.org (https://doi.org/10.15468/dl.rdguv4, 5 August 2020). Picture of M. viridis modified from https://invasions.si.edu/nemesis/species_summary/-47.

CMIST scores for *M. viridis:*

Mean adjusted Likelihood of Invasion: 2.24

Mean adjusted Impact of Invasion: 2.41

Mean adjusted Overall CMIST score: 5.39

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Marenzelleria viridis* is not established in the RA area. However, survey effort is low relative to the size of the RA area. Note: this species has been found elsewhere in the Canadian Arctic although this may have been misidentified (see Q2) as the species complex is taxonomically challenging.

Background Information: There have been no reports found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016), however this species was reported by Brown et al. (2011) in 2005-2006 from samples collected in the Gjoa Haven area of the Arctic Archipelago. Given this is the only known report of this species from the Arctic, it is possible it was a misidentification of the closely related *Marenzelleria arctia* which has recently been found in other locations in the Beaufort Sea, Western Arctic (C. Conlon, Canadian Museum of Nature, personal communication, K. Howland, unpublished data).

Q2- Rate of introduction **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Marenzelleria viridis* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Marenzelleria viridis is present in ports that are directly connected by shipping to Churchill and Deception Bay and could potentially be entrained and transported by these vessels in ballast water (Chan et al., 2012).

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Marenzelleria viridis*. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: From NEMESIS: "*Marenzelleria viridis* is an estuarine spionid polychaete. Juveniles and adults inhabit mudflats and shallow muddy bottoms, usually in areas of variable or low salinity (George, 1966, Atkins et al., 1987, Peterson and Vayssieres, 2010)". (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47)

From NOBANIS: "In shallower areas (<10 m) it prefers sand or gravel bottoms with higher abundance in more densely vegetated areas. Deeper down (>10 m) it is confined to silty clay substrate" (https://www.nobanis.org/globalassets/speciesinfo/m/marenzelleria-neglecta/marenzelleria_neglecta2.pdf for the closely related *M. neglecta*).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Marenzelleria viridis*. It has been found elsewhere in the Arctic indicating it should be able to survive at winter temperatures in the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Marenzelleria viridis is typically found in areas of highly variable salinity (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47). Salinity tolerance is 30-0.5 PSU (Schiedek, 1998) and temperature tolerance 7-23 °C (www.eol.org, according to known distribution). However, species of the same genus have been found in other areas of the Arctic (Bastrop et al., 1997) which would have winter temperatures similar to the risk assessment area.

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are sometimes available for *Marenzelleria viridis* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperatures >=5 °C (upper limit where larval development - the major limiting life stage- can occur) and 10°C (optimal for larval development) resulted in selecting a moderate amount of suitable habitat in the RA area. The species' requirements are well demonstrated and there are reliable environmental data layers for the RA area to show a match.

Background Information: From NEMESIS: "Fecundity of animals depends on salinity, temperature, age and body size - *Marenzelleria* spp. have separate sexes and mature at about 40 mm length (George, 1966, Dauer et al., 1980, Bastrop and Blank, 2006). Females are estimated to produce 10,000 - 46,000 eggs (George, 1966, Bochert, 1997). Adults apparently die after spawning (Atkins et al., 1987). Our picture of the life history of *Marenzelleria* spp. is incomplete, but experimental and field data suggest that abundant populations in oligohaline and tidal fresh waters are maintained by seaward migration and spawning of adults, and the tidal transport of larvae up into estuaries (Dauer et al., 1980), a catadromous life history. The reproduction of *M. viridis* varies depending on geographical location. Studies on a North American and two North Sea populations have shown that these populations spawn in spring but in the enclosed waters of the southern Baltic, spawns in autumn (Bochert and Bick, 1995; and references therein). The differences in the reproductive seasons might be genetically determined (Bochert and Bick, 1995). Gamete production starts whilst temperatures are falling and continues during the coldest months of the year, it is quite possible that a rise in temperature is needed to complete the final ripening of the gametes and to bring on spawning. Indeed, if apparently mature worms are kept at a constant low temperature of approximately 0°C, they can be held for many months without spawning and will eventually reabsorb their sexual products. Conversely, by slowly raising the temperature from 0 to 2°C over a two-week period worms can be made to spawn under laboratory conditions, up to a month before natural spawning (George, 1966). The rate at which larval development proceeds in laboratory cultures is dependent on temperature and salinity: At 20°C metamorphosis is reached in approximately half the time (~26 days) taken at 10°C (~44 days) and at 2°C, no larvae develop beyond the 3- to 4-setiger stage; the rate of development is similar in salinities varying from l0‰ to 30‰, but at 5, growth was greatly slowed and at 2 ‰ it ceased (George, 1966). Bochert et al. (1996), Bochert (1997) found larval development depended on water temperature and salinity and lasted about 4 to 12 weeks - successful larval development from egg to juvenile was not possible below salinities of 5‰, but colonization of oligohaline regions took place by larvae with more than 4 setigers or by swimming juveniles - development was slower at 5°C, than at 10 or

20°C, however developmental abnormalities were noted at the highest temperature of 20°C, suggesting it may be sub-optimal." (https://invasions.si.edu/nemesis/species_summary/-47)

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Marenzelleria viridis.* There are species in the RA area that could act as predators but limited evidence that these can severely limit population growth especially for adults that may be more protected. Although the information is limited, it is on predators specific to the species and known to occur in the RA area.

Background Information: Marenzelleria viridis is eaten by fish, birds and crabs (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47).

From Leppäkoski and Olenin (2000): "In the Baltic Sea, *Macoma balthica* caused a significant mortality of *M. viridis* (Kotta et al., 2001). Although the larvae and young adults of *M. viridis* may be an additional food source for benthic fishes, adults are well protected and not readily available to predators (Bochert, 1997, Schiedek, 1997, Zettler, 1997).

Macoma balthica together with other species of the same genus (*M. calcarea*, *M. loveni*, *M. moesta* and *M. torelli*) are present in the assessment region (Stewart and Lockhart, 2005).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: There is good evidence for widespread natural dispersal with a clear mechanism due to the long pelagic larval phase.

Background Information: Larvae could use selective tidal migration, and/or be transported upstream in saline benthic waters (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47).

In the German part of the Baltic Sea, *M. viridis*' recruitment is confined to inshore waters with a subsequent dispersal of adult specimens, either actively through swimming, or passively via currents (Zettler et al., 1995).

From Leppäkoski and Olenin (2000): "In the Baltic Sea, the approximate (minimum) rate of spread from German Boddens (1985) to Lithuania (1989) was 170 km/year, from there to southern Finland (1990) 480 km/year, and further to northern Quark (1996) 90 km/year. *M. viridis* is reported to migrate by active swimming (cf. Dauer et al., 1980, Bochert et al., 1996).

From Bochert (1997): "The long pelagic phase of *M. viridis* gives the larvae plenty of opportunity to disperse. Like the adults, the larvae possess powerful physiological adaptation mechanisms which enable them to survive stress situations better than other species (Fritzsche and Von Oertzen, 1995, Bochert, 1997). This has enabled *M. viridis* to spread and colonize a wide range of habitats, especially in oligohaline areas."

From Bastrop et al. (1998): *Marenzelleria viridis* produces up to 21 million planktonic larvae per cubic meter of water during its reproductive season, and the planktotrophic larval stage lasts about 8 weeks (George, 1966, Bochert and Bick, 1995).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Marenzelleria viridis* has the capacity for a wide range of anthropogenic dispersal, mainly in ballast, in the RA area. There is solid evidence of anthropogenic dispersal though some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: Marenzelleria viridis has probably been transported as larvae in ballast water (Molnar et al., 2008). Both larvae and benthic juveniles are being transported in the Baltic.

It is one of the most rapidly dispersing non-indigenous species in the Baltic Sea, with first records often made near ports, suggesting introduction via ballast water (Leppäkoski and Olenin, 2000).

From NEMESIS: “From the 1970s to the 1990s, at least three separate contemporaneous cryptic invasions of *Marenzelleria* spp. into the northeastern Atlantic have occurred, probably through ballast water discharges (Sikorski and Bick, 2004, Bastrop and Blank, 2006).” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47).

Ballast transported by domestic coastal resupply and other boat traffic, could spread *M. viridis* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012, Chan et al., 2015). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Marenzelleria viridis* would have a high impact in many areas on population growth in the RA area. There is abundant information showing that this species impacts populations in areas where it has invaded.

Background Information: From NEMESIS: “*Marenzelleria arctia* is invasive in Finland and due to its abundant reproduction there was an increase in biomass of macrozoobenthos there. Because of its invasion, the Gulf of Finland became occupied by almost single-species macrozoobenthos (Maximov, 2011). Due to competition, there was a decline in abundance of the dominant polychaete *Nereis diversicolor* in Danish and Finnish waters (Kotta et al., 2001, Delefosse et al., 2012). In the Baltic Sea, *Marenzelleria* spp. displaced the native deposit-feeding amphipod *Monoporeia affinis* in experiments (Kotta and Ólafsson, 2003).” (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47)

From Kotta et al. (2001) and references therein: “Observations yield some circumstantial evidence that after the invasion of *M. viridis*, densities of the native polychaete *Nereis diversicolor,* the shallow water amphipod *Corophium volutator* and deep-water amphipod *Monoporeia affinis* have dropped considerably.”

From Norwegian Biodiversity Information Center: "Studies from Denmark and the Baltic Sea show that the species affects the habitat by digging very deep into the sediment and can displace local species. In a survey in Odensefjord, Denmark, it was shown that *M. viridis* after establishment led to a significant decline (60% reduction) of the local polychaete *Hediste diversicolor* and displaced it from parts of its original area of occurrence (Delefosse et al., 2012). Similar effects for *H. diversicolor* have been demonstrated in the Baltic Sea for the related (and also introduced) species *M. neglecta* (Didžiulis, 2006), but here, changes in natural environmental conditions (temperature, water exchange) can also occur (Kauppi et al., 2015). *Hediste diversicolor* is a character species on soft-bottom beaches in southern Norway and similar effects can be expected in Denmark. Both species of *Marenzelleria* dig deeper into the sediment than local species and, through bioturbation and increased water supply to deep sediment layers, may affect biogeochemical processes and release of degradation products from the sediments (Quintana et al., 2011, Delefosse et al., 2012, Maximov et al., 2015). Positive effects of this have been demonstrated in the Baltic Sea, but it is more uncertain what significance this has in areas of higher salinity preferred by *M. viridis* (Delefosse et al., 2012). However, there are fears that digging activity may lead to increased mobilization of environmental toxins in areas with highly contaminated bottom sediments (Kauppi et al., 2015, Maximov et al., 2015). Based on available knowledge, *M. viridis* is considered to have a medium ecological effect.

Marenzelleria viridis is considered to be very high risk in Norway because it has large invasion potential while affecting native species as well as the natural state of digging activity where it establishes. In Norway, the species can be expected to colonize shallow soft bottom areas in brackish water in a similar manner to that around the North Sea. Brackish water systems are generally poor in species, but it can be expected that the effects in Norway can be very different from place to place." (https://artsdatabanken.no/fremmedarter/2018/N/2682)

From NEMESIS: "The invasion of *M. viridis* and *M. neglecta* in Danish and Finnish waters is associated with a sharp decline in abundance of the dominant polychaete *Hediste* (=*Nereis*) *diversicolor*, which may have been due to competition (Kotta et al., 2001, Delefosse et al., 2012). In the Baltic (Asko, Finland), *Marenzelleria* spp. displaced the native deposit-feeding amphipod *Monoporeia affinis* in experiments (Kotta and Ólafsson, 2003). However, *Marenzelleria* is out-competed by the native bivalve *Macoma balthica* and does not successfully invade *Macoma*-dominated communities (Kotta et al., 2001).

Changes in sediment properties and communities due to *Marenzelleria* spp. have been reported from European waters. The adults of *M. viridis* and M. *neglecta* create unbranched burrows down to 25-35 cm in the sediment (Atkins et al., 1987, Hietanen et al., 2007, Renz and Forster, 2013). Burrow structures and sediment impacts of *M. viridis* and *M. neglecta* are similar (Renz and Forster, 2013). Dense populations of adult worms rework the sediment, bringing buried organic materials and nutrients to the surface, possibly increasing fluxes of NH^{4+} and P to the water column initially, but possibly promoting phosphorus retention and nitrification in the longer term (Hietanen et al., 2007, Hedman et al., 2011, Norkko et al., 2012). Experimental studies with worms and sediments from Odense Fjord, Denmark, showed that the deeper-burrowing introduced *M. viridis* increased sulfur reduction and H_2S in pore water, compared to the native *Hediste diversicolor*, favoring more sulfide-tolerant species (Kristensen et al., 2011). In the presence of *Marenzelleria* sp., the polychaete *Hediste diversicolor* Atkins et al. (1987)- Tay estuary, Scotland; Kotta et al. (2001)- Baltic, Finland) and a community of oligochaetes and chironomids Żmudziński (1996), Vistula Lagoon, Poland) sharply declined, possibly due to competition and habitat change."
https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Marenzelleria viridis* would have high impact in many areas on communities in the RA area. There is abundant information showing that this species impacts a number of communities in areas where it has invaded.

Background Information: From Leppäkoski and Olenin (2000): *Marenzelleria viridis* introduces competitive interactions for food and space or both with native benthic species (Kotta et al., 2001). Changes in sediment properties and communities due to *Marenzelleria* spp. have also been observed. Native oligochaetes originally inhabited the uppermost 5 cm layer, but due to the porosity caused by *M. viridis*, they now extend to 15 cm (Żmudziński, 1996).

From Norwegian Biodiversity Information Center: "Studies from Denmark and the Baltic Sea show that the species affects the habitat by digging very deep into the sediment and can displace local species. In a survey in Odense Fjord, Denmark, it was shown that *M. viridis* after establishment led to a significant decline (60% reduction) of the local polychaete *Hediste diversicolor* and displaced it from parts of its original area of occurrence (Delefosse et al., 2012). Similar effects for *H. diversicolor* have been demonstrated in the Baltic Sea for the related (and also introduced) species *M. neglecta* (Didžiulis, 2006), but here, changes in natural environmental conditions (temperature, water exchange) can also occur (Kauppi et al., 2015). *Hediste diversicolor* is a character species on soft-bottom beaches in southern Norway and similar effects that can be expected in Denmark. Both species of *Marenzelleria* dig deeper into the sediment than local species and, through bioturbation and increased water supply to deep sediment layers, may affect biogeochemical processes and release of degradation products from the sediments (Quintana et al., 2011, Delefosse et al., 2012, Maximov et al., 2015). Positive effects of this have been demonstrated in the Baltic Sea, but it is more uncertain what significance this has in areas of higher salinity preferred by *M. viridis* (Delefosse et al., 2012). At the same time, there are fears that digging activity may lead to increased mobilization of environmental toxins in areas with highly contaminated bottom sediments (Kauppi et al., 2015, Maximov et al., 2015). Based on available knowledge, *M. viridis* is considered to have a medium ecological effect.

Marenzelleria viridis is considered to be very high risk in Norway because it has a large invasion potential while affecting native species and affecting the natural state of digging activity where it establishes. In Norway, the species can be expected to colonize shallow soft bottom areas in brackish water in a similar manner to that around the North Sea. Brackish water systems are generally poor in species, but it can be expected that the effects in Norway can be very different from place to place." (https://artsdatabanken.no/fremmedarter/2018/N/2682)

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Changes in sediment properties and communities due to *Marenzelleria* spp. have been reported from European waters. The adults of *M. viridis* and M*. neglecta* create unbranched burrows down to 25-35 cm in the sediment (Atkins et al., 1987, Hietanen et al., 2007, Renz and Forster, 2013). Burrow structures and sediment impacts of *M. viridis* and *M. neglecta* are similar (Renz and Forster, 2013). Dense populations of adult worms rework the sediment, bringing buried organic materials and nutrients to the surface, possibly increasing fluxes of NH^{4+} and P to the water column initially, but possibly promoting phosphorus retention and nitrification in the longer term (Hietanen et al., 2007, Hedman et al., 2011, Norkko et al., 2012). Experimental studies with worms and sediments from Odense Fjord, Denmark, showed that the deeper-burrowing introduced *M. viridis* increased sulfur reduction and H_2S in pore water, compared to the native *Hediste diversicolor*, favoring more sulfide-tolerant species (Kristensen et al., 2011). In the presence of *Marenzelleria* sp., the polychaete *Hediste diversicolor* Atkins et al. (1987)- Tay estuary, Scotland; Kotta et al. (2001)- Baltic, Finland) and a community of oligochaetes and chironomids Żmudziński (1996), Vistula Lagoon, Poland) sharply declined, possibly due to competition and habitat change."
https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Marenzelleria viridis* would have high impact in many areas on habitat in the RA area. There is abundant information showing that this species may impact benthic habitat to some degree in a number of ways.

Background Information: From NEMESIS*:* "Changes in sediment properties and communities due to Marenzelleria spp. The adults create unbranched burrows down to 25-35 cm in the sediment (Atkins et al., 1987, Hietanen et al., 2007, Renz and Forster, 2013). Dense populations of adult worms rework the sediment, bringing buried organic materials and nutrients to the surface, possibly increasing fluxes of NH^{4+} and P to the water column initially, but possibly promoting phosphorus retention and nitrification in the longer term (Hietanen et al., 2007, Hedman et al., 2011, Norkko et al., 2012).

Changes in sediment properties and communities due to *Marenzelleria* spp. have been reported from European waters. The adults of *M. viridis* and *M. neglecta* create unbranched burrows down to 25-35 cm in the sediment (Atkins et al., 1987, Hietanen et al., 2007, Renz and Forster, 2013). Burrow structures and sediment impacts of *M. viridis* and *M. neglecta* are similar, while *M. arctia* digs shallower U-shaped burrows, and has less impact on sediments (Renz and Forster, 2013). Dense populations of adult worms rework the sediment, bringing buried organic materials and nutrients to the surface, possibly increasing fluxes of NH^{4+} and P to the water column initially, but possibly promoting phosphorus retention and nitrification in the longer term (Hietanen et al., 2007, Hedman et al., 2011, Norkko et al., 2012). Experimental studies with worms and sediments from Odense Fjord, Denmark, showed that the deeper-burrowing introduced *M. viridis* increased sulfur reduction and H_2S in pore water, compared to the native *Hediste diversicolor*, favoring more sulfide-tolerant species (Kristensen et al., 2011). In the presence of *Marenzelleria* sp., the polychaete *Hediste diversicolor* Atkins et al. (1987)- Tay estuary, Scotland; Kotta et al. (2001)- Baltic, Finland) and a community of oligochaetes and chironomids Żmudziński (1996), Vistula Lagoon, Poland) sharply declined, possibly due to competition and habitat change."
https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47

From Norwegian Biodiversity Information Center: "*M. viridis* is considered to be very high risk in Norway because it has a large invasion potential while affecting native species and affecting the natural state of digging activity where it establishes. In Norway, the species can be expected to colonize shallow soft bottom areas in brackish water in a similar manner to that around the North Sea. Brackish water systems are generally poor in species, but it can be expected that the effects in Norway can be very different from place to place." (https://artsdatabanken.no/fremmedarter/2018/N/2682)

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Marenzelleria viridis* would have high impacts in many areas on ecosystem function in the RA area. There is abundant information showing that this species impacts ecosystem function to some degree in a number of ways.

Background Information: These polychaetes are characterized by a considerable bioturbation and bioirrigation activity, giving the deep-water areas a new functional group of benthos as it happened with *Marenzelleria arctia* in the Gulf of Finland. This will also lead to a reorganization at the scale of the entire ecosystem due to changes and impact on the biogeochemical process and trophic relations (Maximov, 2011). The presence of *M. viridis* increased benthic production (Chlorophyll *a*) in the Baltic Sea (Kotta et al., 2001). Higher sediment Chl *a* content indicates higher biodeposition and/or bioturbating activity (Kotta et al., 2001). This can accelerate nitrogen remineralisation and transformation process within the sediment (Pelegri and Blackburn, 1995, Kotta et al., 2001). Being a deep burrowing deposit feeder, it introduced a new functional group in the northern Baltic Sea (https://www.nobanis.org/globalassets/speciesinfo/m/marenzelleria-neglecta/marenzelleria_neglecta2.pdf). *Marenzelleria viridis* is reported to migrate by active swimming (cf. Dauer et al., 1980, Bochert et al., 1996). This movement may serve as an important link in energy transfer between pelagic and benthic subsystems. As a result, *M. viridis* mobilizes organic matter deposited in deeper sediment strata, links benthic and pelagic subsystems, and creates new microhabitats for associated fauna (Leppäkoski and Olenin, 2000).

From NEMESIS: "*Marenzelleria viridis* has reached high densities in the Sacramento-San Joaquin Delta (Cohen et al., 1995, Peterson and Vayssieres, 2010), but impacts have not been extensively studied. In European waters, *M. viridis*, together with *M. neglecta* and *M. arctia*, have become dominant organisms in benthic communities, partially replacing native infauna, and affecting the characteristics of sediments and their communities (Atkins et al., 1987, Hietanen et al., 2007, Hedman et al., 2011). This polychaete is a potential food source for fishes, but no economic impacts have been reported.

Changes in sediment properties and communities due to *Marenzelleria* spp. have been reported from European waters. The adults of *M. viridis* and M*. neglecta* create unbranched burrows down to 25-35 cm in the sediment (Atkins et al., 1987, Hietanen et al., 2007, Renz and Forster, 2013). Dense populations of adult worms rework the sediment, bringing buried organic materials and nutrients to the surface, possibly increasing fluxes of NH^{4+} and P to the water column initially, but possibly promoting phosphorus retention and nitrification in the longer term (Hietanen et al., 2007, Hedman et al., 2011, Norkko et al., 2012). Experimental studies with worms and sediments from Odense Fjord, Denmark, showed that the deeper-burrowing introduced *M. viridis* increased sulfur reduction and H_2S in pore water, compared to the native *Hediste diversicolor*, favoring more sulfide-tolerant species (Kristensen et al., 2011). In the presence of *Marenzelleria* sp., the polychaete *Hediste diversicolor* Atkins et al. (1987)- Tay estuary, Scotland; Kotta et al. (2001)- Baltic, Finland) and a community of oligochaetes and chironomids Żmudziński (1996), Vistula Lagoon, Poland) sharply declined, possibly due to competition and habitat change." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47)

From Norwegian Biodiversity Information Center: "*M. viridis* is considered to be very high risk in Norway because it has a large invasion potential while affecting native species and affecting the natural state of digging activity where it establishes. In Norway, the species can be expected to colonize shallow soft bottom areas in brackish water in a similar manner to that around the North Sea. Brackish water systems are generally poor in species, but it can be expected that the effects in Norway can be very different from place to place." (https://artsdatabanken.no/fremmedarter/2018/N/2682)

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Marenzelleria viridis* would have low or no impact associated with diseases, parasites or travelers in the RA area. There is little information available and no evidence of impacts from disease/parasites that could affect native species in the RA area.

Background Information: From Phelan et al. (2016) and references therein: "confirmed the presence of three species of trematodes (*Himasthla quissetensis*, *Lepocreadium setiferoides*, and *Zoogonus lasius*) in *M. viridis* as a host. Trematodes use *M. viridis* as a second intermediate host and this intermediate host demonstrates a clear immune response to metacercarial infection. A variety of vertebrates (e.g., fish, birds, and mammals) serve as definitive hosts for trematode species. (Within gastropod hosts, trematodes replace host tissue (especially in the gonad region, resulting in castration) and may occupy a significant portion of the shell volume. After developing as sporocysts and/or rediae and multiplying asexually, they emerge from the first intermediate host as free-swimming cercariae and seek a second intermediate host, which may be polychaetes or other invertebrate or vertebrate taxa. Within second intermediate hosts, they encyst as metacercariae until they are ingested by a definitive host. Trematodes complete their life cycle by reproducing sexually and releasing eggs through the host's feces. Trematodes typically do not kill their hosts but can affect their behavior, overall fitness, and, scaled up to the community level, the structure of coastal systems."

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Marenzelleria viridis* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native polychaetes of the same genus in the RA area, but no evidence of hybridization found in the literature.

Background Information: The genus *Marenzelleria* consists of five species, which are very difficult to discriminate by morphological characters alone (Blank et al., 2008). Sibling species are: *M. viridis*, *M. neglecta*, and *M. arctia* (the latter of which has Arctic Ocean origin, and is abundant in the inner Baltic (Bastrop and Blank, 2006) and may have a different life history (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47)). Morphologically identical species can only be distinguished from each other by considering reproductive characteristics (Bochert and Bick, 1995).

There are no sibling species in the RA area according to OBIS, GBIF, and EoL, however *M. arctia* has been found in Churchill (Howland, unpublished) and other areas of the Canadian Arctic (C. Conlon, pers. Comm.).

Q15- Impact on at-risk species (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Marenzelleria viridis* would have low or no impact on at-risk or depleted species in the RA area. Species at risk do not overlap with the species habitat (mostly estuarine, intertidal and nearshore).

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through food web and ecosystem impacts – these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; thorny skate, *Amblyraja radiata*. Walrus populations of risk/special concern may also be impacted through food web changes in benthic habitats as they feed on benthic invertebrates. However, this may not be of relevance as *M. viridis* is mostly present in estuaries and fresh waters.

Q16- Impact on fisheries (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Marenzelleria viridis* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. There may be slight impacts to fisheries species. Aquaculture is not practiced in the area.

Background Information: Through the infestation with *Lepocreadium setiferoides* trematode, the sculpin *Myoxocephalus octodecemspinosus* could be affected which is a species harvested by Inuit (Stewart and Lockhart, 2005). Likewise, shallow water benthic species could be impacted by competition/food web effects and/or habitat/ecosystem effect (Kotta et al., 2001, Kotta and Ólafsson, 2003). *Mya* spp., as well as Mussels (*Mytilus* spp.), sea urchin (*Strongylocentrotus droebachiensis*), brown sea cucumber (*Cucumaria japonica*), and Iceland scallops (*Chlamys islandica*) are harvested by Inuit in the risk assessment area and could be impacted (Stewart and Lockhart, 2005; Igloolik Hunters and Trappers Organization, personal communication). However, the parasite does not have the 1st intermediate host in the area and thus, the life cycle may not be completed.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Marenzelleria viridis* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: It is native to the East Coast of North America from Nova Scotia to Delaware and probably Newfoundland to Chesapeake Bay; although further sampling is needed since there are at least two additional cryptic species in this area (*M. neglecta* and *M. bastropi*). It is introduced in California, Scotland, the North Sea, and the Baltic Sea (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-47). The genus

Marenzelleria consists of five species, which are very difficult to discriminate by morphological characters alone (Blank et al., 2008). Originally inhabiting only North American estuaries, the polychaete *M. viridis* (Verrill, 1873) has been found in various North Sea (Essink and Kleef, 1988, McLusky et al., 1993) and Baltic estuaries (Bick and Burckhardt, 1989) since the mid-eighties (Bochert and Bick, 1995).

Membranipora membranacea **(Linnaeus, 1767)**

Phylum: Bryozoa

Class: Gymnolaemata

Order: Cheilostomatida

Family: Membraniporidae

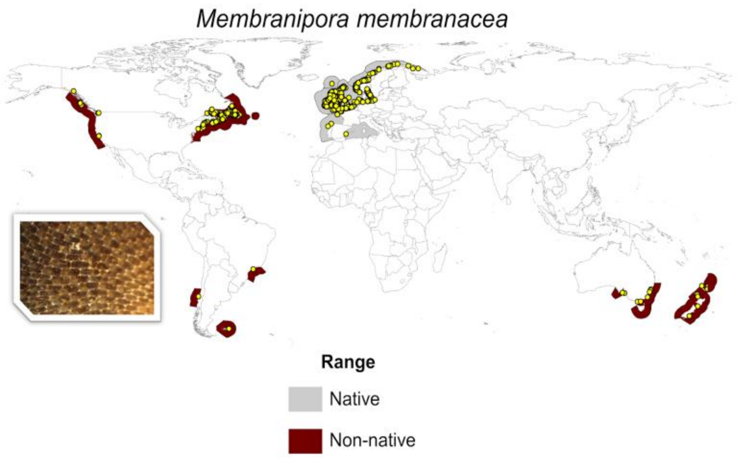

Figure 22: Ecoregions where Membranipora membranacea is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Membranipora membranacea occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://api.gbif.org/v0.9/occurrence/download/request/0009142-140429114108248.zip, 10 June 2014), and NEMESIS (http://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=155824). Picture of M. membranacea modified from https://invasions.si.edu/nemesis/species_summary/155824. Note that some studies on molecular phylogeny related to bryozoans have shown that this species is composed by a complex of multiple clades that were originated from the North Pacific, and that the only verified invasion is in the Northwest Atlantic (https://invasions.si.edu/nemesis/species_summary/155824, Schwaninger, 2008).

CMIST scores for *M. membranacea:*

Mean adjusted Likelihood of Invasion: 2.12

Mean adjusted Impact of Invasion: 2.46

Mean adjusted Overall CMIST score: 5.22

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Membranipora membranacea* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports of the species presence were found from Hudson Bay (Stewart and Lockhart, 2005, Goldsmit et al., 2014, Brown et al., 2016, Chain et al., 2016, Goldsmit, 2016). *Membranipora membranacea* is established in the Gulf of St. Lawrence, Quebec and Nova Scotia. It was first recorded in Mahone Bay and St. Margaret's Bay in Nova Scotia in 1992 (Burridge et al., 2012), has been present since the early 2000s in western Newfoundland and southern Labrador (Caines and Gagnon, 2012), and is also invading Prince Edward Island (Vercaemer et al., 2012).

Q2- Rate of introduction (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Membranipora membranacea* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread and abundant in connected ecoregions and there is at least one vector for transport.

Background Information: Propagule pressure is unknown but *M. membranacea* is present at ports directly connected to Churchill and Deception Bay by shipping and could be entrained and transported by these vessels in ballast water, or on fouled hulls (Chan et al., 2012). This species was found attached to algae in the sea chests and/or strainers of 2 of 14 ocean vessels sampled at ports near the mouth of the St. Lawrence River (Couture and Simard, 2007).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Membranipora membranacea.* There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the habitat classification.

Background Information: Colonies are typically found on kelps (e.g., *Agarum clathratum*, *Laminaria digitata*, and *Saccharina latissima*) and fucoids (e.g., *Fucus evanescens*) from the intertidal zone to a depth of 12 m (Saunders and Metaxas, 2007, 2009a). In Nova Scotia, *M. membranacea* coverage on kelp was generally 0% in late spring, increased significantly in August and September, peaked in October through December, then declined during the winter (Saunders and Metaxas, 2009b). More overwintering occurred on the kelp species *L. digitata* and *A. clathratum* than on *S. latissima*. *Fucus* spp. provide an important refuge for overwintering colonies, particularly where defoliation of kelps has been extensive, and characteristics of this substratum probably facilitate early reproduction and local spread (Yorke and Metaxas, 2012). These overwintering colonies likely provide larvae for the first settlement in May and June (Saunders and Metaxas, 2007).

In California, *M. membranacea* feeding was most successful at ambient flow speeds of 10-12 cm/s (Arkema, 2009). In the Gulf of Maine the species was more common at exposed than protected sites (Berman et al., 1992), perhaps due to differences in current that affect feeding. In California larvae can occur from the surface to a depth of at least 60 m (Yoshioka, 1982).

Kelp beds are present along the coasts of the HB LME (Stewart and Lockhart, 2005), particularly in areas with some shoreline development to provide protection from storms and ice scour. The overall distribution, species composition, abundance, and continuity of kelp beds in the region is unknown.

Q4- Survival (suitable environment) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest the RA area offers a moderate proportion of suitable environmental conditions for *Membranipora membranacea.* The species can survive under ice cover at sub-zero temperatures in other locations (e.g., Gulf of St.

Lawrence), but these are at the extreme northern extent of its distribution where winters would be shorter and ice cover thinner than in most of the RA area. The species' requirements are well documented, and existing modelling studies suggest there is a limited amount of suitable habitat in the RA area.

Background Information: Sea temperature is a significant determinant of settlement, recruitment, and colony cover of *M. membranacea* throughout its invaded range in the northwest Atlantic (Newfoundland and Labrador: Caines and Gagnon, 2012). At Red Bay, Labrador, the site of the current northernmost record, the species inhabited waters that averaged 6.6°C during the warmest half of the year (June–November; range ca. 2.5-14°C, depth 3.5 m), with winter averages of (-1.7 –2.0°C) (Caines and Gagnon, 2012). Low settlement and recruitment suggest the species is near its thermal limits at this site; temperature tolerances lower than those of native *M. membranacea* in Alaska suggest that the species is adapting to colder water as it invades shallow coastal habitats further north in the northwestern Atlantic. Zooids have an upper LD50-24h of 25.3°C (Menon, 1972). The addition of zooids by budding at the colony rim takes 29 days at 6°C, 10 days at 12°C, and 6 days at 18°C.

Membranipora membranacea have established in St. Margaret's Bay, NS waters with daily averaged temperatures at 4 m depth ranging from 20.92°C to 1.2°C (-0.69°C at 8 m) (Saunders and Metaxas, 2007, Denley and Metaxas, 2015). Mean monthly temperatures in the region at <10 m depth peaks in August/September at ca. 15.5°C and is coldest in February/March at about 0.6°C (Scheibling and Gagnon, 2009). Between 3 and 9°C the water temperature in laboratory studies had little effect on mortality rate (Denley and Metaxas, 2016). Between ca. 6 and 18°C, growth rate and colony size are positively related to temperature, in the lab and field and in its native range (Menon, 1972), and in Nova Scotia (Saunders and Metaxas, 2009b). In Nova Scotia, larval abundance was higher in the warmer, fresher water above the pycnocline than in the colder, more saline water below (Saunders and Metaxas, 2010). Data on adult salinity tolerances are few, with a minimum salinity tolerance reported from New Hampshire of 27 PSU (Blezard, 1999). Niche modelling results showed that projected environmental conditions under global warming will include a moderate region of the assessment area as suitable habitat (Goldsmit et al., 2018).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest reproductive requirements are sometimes available for *Membranipora membranacea* in the RA area. ArcGIS calculations identified a moderate amount of habitat available in the RA area with maximum sea surface temperature >=5°C, the lowest limit above which colonies have been experimentally shown to grow and settlement in the field has been observed. Although there are reliable environmental data layers for the RA area to show a match, there is limited information on requirements for sexual reproduction. Settlers have been observed at temperatures of <5°C, suggesting the possibility of recruitment at these lower temperatures, but there is uncertainty regarding temperatures at which they were produced and regarding their abilities to establish colonies.

Background Information: Membranipora membranacea reproduce asexually by budding at the periphery to form sheet-like colonies (Berman et al., 1992). They can also reproduce sexually when crowded by conspecifics. Fertilized eggs are released into the water column, not brooded like most other bryozoans (Yoshioka, 1982). In California waters the planktonic larvae (cyphonautes), remain in the water column for about 4 weeks but larvae can be maintained in the laboratory for up to 8 weeks. Based on other species, the larval duration is likely inversely related to temperature (i.e., longer in colder temperatures) but no information was found on larval duration in cold water. In Nova Scotia larval settlement began in May and peaked in September and October (Saunders and Metaxas, 2007). Settlers were present but at low abundance in December to April when water temperatures were <5 °C (Saunders and Metaxas, 2007). Their ability to develop into colonies at those temperatures is unknown. *Membranipora membranacea* larvae show preference for the younger, proximal regions of kelp blades when settling, which suggests they can detect differences in habitat quality at the scale of a single kelp blade (Denley et al., 2014). In native populations in Washington, colonies are usually reproductive 40 days after settlement (Harvell and Helling, 1993). Experimental research on effects of low water temperature on growth and survival of larvae, settlers, and colonies is required to improve our ability to predict changes in populations of *M. membranacea*; also on the factors triggering settlement (Caines and Gagnon, 2012). Saunders and Metaxas (2009b) observed colonies to grow in the laboratory at 6 °C under both food-limited and unlimited conditions.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Membranipora membranacea.* Several studies provide evidence of predation on this species from taxa that could occur in the RA area and slow population growth. Most information, although extensive, is general and extrapolated to species in the RA area.

Background Information: In its native range along the Pacific coast of North America, *M. membranacea* is kept in check by the grazing of specialist nudibranchs and fish (Yoshioka, 1982). In its introduced range the species has benefited from lack of predators and few competitors (Burridge et al., 2012). There are a few predators of *M. membranacea* in the northwestern Atlantic, including fish, the snail *Astyris lunata* (Say), and nudibranchs *Onchidoris muricata* (Müller) and *Doridella obscura* (Verrill), but none has demonstrated the ability to control populations of this invader (Chapman et al., 2002, Caines and Gagnon, 2012; and references therein). In NS, competitive standoffs that preserve native diversity can occur between the native bryozoan *Electra pilosa* and invasive *M. membranacea* on fucoid algae (Yorke and Metaxas, 2011).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* has the capacity for a wide range of natural dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal based on their extended planktonic larval stage and history of spread.

Background Information: Unlike many other bryozoans, *M. membranacea* releases its fertilized eggs into the water column (Yoshioka, 1982). In California waters the planktonic larvae (cyphonautes), remain in the water column for about 4 weeks but larvae can be maintained in the laboratory for up to 8 weeks. No information was found on larval duration in cold water but it is likely >4 weeks. The larvae are relatively weak swimmers, using only cilia for directed movement (Saunders and Metaxas, 2010). In NS they tend to be more abundant in the warmer, fresher surface waters, so their dispersal depends largely upon the surface currents and duration of the larval period (Saunders and Metaxas, 2010); they avoid surface waters that are warmer than ca. 23 °C (Yoshioka, 1982). The species' relatively continuous distribution in NS (i.e., wherever kelps are present) suggests it has been spread by natural dispersal of the planktonic larvae (Watanabe et al., 2010). Adult dispersal can occur by rafting on dislodged kelp (Schwaninger, 2008) and drift plastic (Aliani and Molcard, 2003). Northward spread of *M. membranacea* introduced into the Northwest Atlantic has occurred rapidly since 1987 and is ongoing (Burridge et al. 2012, Denley et al., 2019). Recent experimental results suggest that it is not limited by temperature but rather the availability of kelp substrate (Denley et al., 2019).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is good potential and supporting evidence for wide range dispersal based on ballast and biofouling through shipping and boating.

Background Information: Membranipora membranacea can be entrained and transported by vessels in ballast water, or on fouled hulls (Chan et al., 2012). Simard et al. (2011) found the dispersive larvae in ballast water of ships in eastern Canadian ports. This species also encrusts on buoys (Woods Hole Oceanographic and United States, 1952), piers and panels (Dijkstra and Harris, 2009), kelp (Schwaninger, 2008) and debris (Aliani and Molcard, 2003). Adults could be transported attached to vessel surfaces (Burridge et al., 2012). It has been found attached to kelp in sea chests and strainers of ocean-going vessels at ports near the mouth of the St. Lawrence River (Couture and Simard, 2007).

Domestic coastal resupply and other boat traffic could spread *M. membranacea* via ballast or biofouling from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggest that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread. Adults could be transported on fouled hulls, and fertilized eggs, pelagic larvae, or colony fragments in ballast water. There is no aquaculture in the region at present to attract *M. membranacea* that could be transported with fouled equipment.

Q9- Impact on population **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* would have high impacts in many areas on population growth in the RA area. There is clear evidence of major impacts on various related and unrelated taxa and communities in invaded coastal zones.

Background Information: Membranipora membranacea colonizes benthic macroalgae but is particularly abundant on kelps of the genus *Laminaria* (Berman et al., 1992). Heavy settlement reduces kelp survival and biomass, and favours dominance by algae that provide less cover (Fofonoff et al., 2003). In the Gulf of Maine, large blades of these kelps were more heavily encrusted than small ones, and kelps from exposed sites were more infested than conspecifics from protected sites. These encrustations may affect kelp nutrient metabolism (Hurd et al., 1994) and photosynthesis (Molina et al., 1991). The kelp is unlikely to benefit from nitrogen excreted by *M. membranacea* (Hurd et al., 2000, Hepburn et al., 2012).

Invasive *M. membranacea* is of particular ecological significance in the northwest Atlantic, where it is the dominant epiphyte on laminarian kelps and the main driver for the defoliation of kelp beds (Denley and Metaxas, 2016). Large-scale defoliation of kelp beds by *M. membranacea* in New England (Lambert et al., 1992) and Nova Scotia (Scheibling et al., 1999) has been attributed primarily to a change in flexibility and tissue strength of encrusted fronds, which increases fragmentation rate during wave surges and storm disturbance (Chapman et al., 2002). Localized growth tissues are often lost with fragmenting blades, precluding regrowth, but juvenile kelps will fill in over time.

Introduced *M. membranacea* in the Gulf of Maine overgrow the native bryozoans *Obelia geniculata* and *Electra pilosa* in the overwhelming majority of encounters on kelp but these native species are more common on other algal hosts, so competitive dominance by *M. membranacea* is unlikely (Berman et al., 1992). The gastropod *Lacuna vincta*, which grazes on *Laminaria*, avoids *M. membranacea* cover and by concentrating its grazing on non-encrusted areas of blades may act synergistically with these bryozoans to increase canopy loss (O'Brien et al., 2013).

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* would have high impact in many areas on communities in the RA area. There is clear evidence of major impacts on various related and unrelated taxa and communities in invaded coastal zones.

Background Information: Membranipora membranacea settles on and damages a variety of brown algae, including *Saccharina latissima*, *Laminaria digitata*, *Fucus evanescens* and *Desmarestia aculeata* which occur in the HB LME (Bell and Macfarlane, 1933). Recurrent seasonal outbreaks of this invasive bryozoan have had a devastating effect on native kelp populations in Nova Scotia, facilitating establishment and growth of the invasive green alga *Codium fragile* ssp. *fragile* (Scheibling and Gagnon, 2009). Changes in the cover of *M. membranacea* on kelp, and in the cover of kelp on the seabed, are reciprocal and seasonal (Scheibling and Gagnon, 2009). Thermal history during the summer/fall period of bryozoan colony growth explains a large proportion (83%) of the interannual variation in peak cover of *M. membranacea* on kelp. Annual decreases in kelp cover and blade size are related to the degree of infestation by *M. membranacea*, and not to wave action alone. Because *M. membranacea* does not cause extensive defoliation of non-laminarian seaweeds, this may limit the establishment of *C. fragile* in algal beds not dominated by kelp (Watanabe et al., 2010). Differences in the abundance of this introduced species among its algal hosts arise as a result of both differential settlement and survival (Saunders and Metaxas, 2009a). In turn, differential susceptibility among hosts to infestation may result in alteration of the algal community structure.

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* would have high impact on habitat in many areas of the RA area. There is clear evidence of major impacts on kelp communities that alter habitats in invaded coastal zones.

Background Information: Kelp beds are prime habitat for many invertebrates, fishes, and migratory birds. Heavy settlement reduces kelp survival and biomass, and favours dominance by algae that provide less cover (Fofonoff et al., 2003). Outbreaks of *M. membranacea* in the northwest Atlantic are the result of periods of early and high settlement and recruitment (Saunders and Metaxas, 2008). By causing kelp defoliation, these outbreaks can have a pronounced ecological effect on kelp beds. Where *Codium fragile* is present it may replace the kelp but it does not prevent re-colonization, at a decadal level, by native kelps (Watanabe et al., 2010).

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* would have high impact in many areas on ecosystem function in the RA area. There is clear evidence of major impacts on kelp communities that alter ecosystem function in invaded coastal zones.

Background Information: Loss of kelp canopy, caused by *Membranipora membranacea* encrustation and the attendant replacement of kelp beds by meadows of the invasive green algae *Codium fragile*, have had profound effects on habitat structure and benthic community organization along the Atlantic coast of North America (Harris and Tyrrell, 2001, Chapman et al., 2002, Levin et al., 2002, Fofonoff et al., 2003). Increased production of kelp detritus during outbreaks of *M. membranacea* in the shallow subtidal zone also affects energy flow to adjacent ecosystems in deeper waters (Scheibling and Gagnon, 2009). In Mahone Bay, NS, *M. membranacea* became the dominant epiphyte on kelps within two years of its introduction (Berman et al., 1992).

Q13- Associated diseases, parasites, and travellers (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Membranipora membranacea* would have low or no impact associated with diseases, parasites or travelers in the RA area.

Background Information: No reports were found of parasites or diseases that may be introduced with *M. membranacea. Loxosomella nordgaardi*, a non-colonial Entoproctan (Family Loxosomatidae), inhabit some bryozoans (Order Cheilostomata) on surfaces where there are small water currents (Yakovis 2002). Some of these bryozoans belong to the same order as *M. membranacea*. Whether *L. nordgaardi* is likely to be newly introduced with *M. membranacea* and impact species within the RA area is uncertain.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* would have low or no genetic impact on other species in the RA area. There are no species of the same genus in the RA area.

Background Information: Serratiflustra serrulata (formerly *Membranipora serrulata*) is present in the HB LME (Stewart and Lockhart, 2005) but hybridization is unlikely.

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Membranipora membranacea* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There is potential for major impacts on kelp communities in coastal zones. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay; at-risk fish, bird, and mammal species are unlikely to be affected. Deterioration of kelp forests would be of concern as it could have cascading effects and affect migratory waterfowl.

Q16- Impact on fisheries (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Membranipora membranacea* would have high impact in many areas on fished species in the RA area. There is potential for major impacts on kelp and associated communities, including scallops and urchins in coastal zones.

Background Information: Kelp is harvested from the HB LME for subsistence, and has been harvested in larger quantities (35 tonnes) for a commercial test fishery at Whale Cove (Stewart and Lockhart, 2005). Fouling by *M. membranacea* reduces the quality and value of kelp for commercial harvest (Førde et al., 2016). Gonadic and somatic growth of the green sea urchin *Strongylocentrotus droebachiensis* can be reduced when defoliation of kelp beds forces them to forage on species with lower nutritional value (Lyons and Scheibling, 2007). These urchins are harvested from the HB LME for subsistence and have been harvested by commercial test fisheries in the Belcher islands (Stewart and Lockhart, 2005). *S. droebachiensis* is an important food of harvested species such as the Hudson Bay subspecies of the common eider duck (*Somateria mollissima sedentaria*) (Gilchrist et al., 2006).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Membranipora membranacea* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Membranipora membranacea has been introduced to the Atlantic coast of North America (USA, Canada) from the eastern Atlantic (western Europe) (Berman et al., 1992, Schwaninger, 2008, Burridge et al., 2012; see also Fofonoff et al., 2003) and references therein). This spatially disjunct and cryptic species, or species complex, consists of 3 long-separated genetic clades in the North Pacific, and monophyletic clades in the North Atlantic, Southeast Pacific (Chile), Southwest Pacific (Australia, New Zealand), and Southeast Atlantic (South Africa). The plasticity and paucity of identifying characteristics in *Membranipora* makes its taxonomy difficult. Until recently some of these clades were separated on the basis of morphology into different species, but these morphs proved to be genetically indistinguishable at the allozyme level. Modelling by (Saunders and Metaxas, 2010) suggests that warmer winters and springs will result in colonies of *M. membranacea* occurring earlier and in higher abundance in the western North Atlantic, and that warmer temperatures during the growth period will increase colony coverage on kelps. If water temperatures in the coastal western North Atlantic warm by as little as 1 °C, which is within the range predicted to occur with climate change, outbreaks of *M. membranacea* will increase in frequency and intensity. Due to the associated loss of kelp, and the potential for defoliated regions to become populated by other nonindigenous species, this could have significant implications for rocky subtidal ecosystems.

PHYTOBENTHOS – MACROALGAE

Codium fragile subsp*. fragile* (Suringar) Hariot, 1889

Phylum: Chlorophyta
Class: Ulvophyceae
Order: Bryopsidales
Family: Codiacea

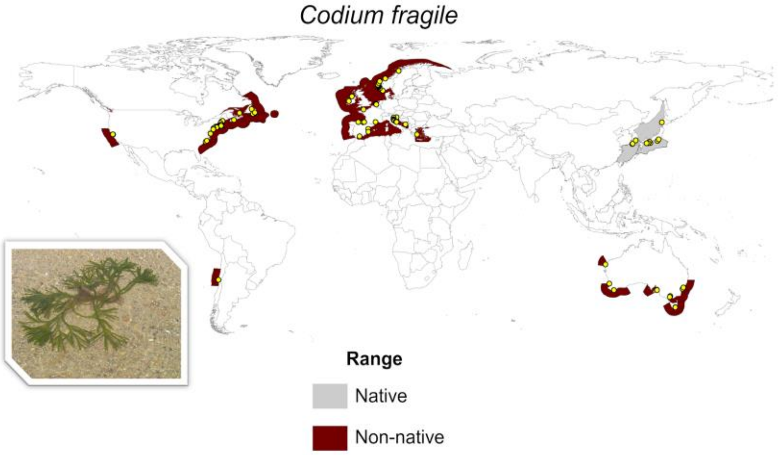

Figure 23: Ecoregions where Codium fragile subsp. fragile is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Codium fragile occurrence points were obtained from GBIF.org (http://doi.org/10.15468/dl.biwzum, 3 May 2017), McDonald et al. (2015), Armitage et al. (2014), Madariaga et al. (2014), Josselyn and West (1985), Carlton and Scanlon (1985), Bulleri and Airoldi (2005), Chavanich et al. (2006), Drouin et al. (2012), Mathieson et al. (2008b), Scheibling and Gagnon (2006), Provan et al. (2005), and Matheson et al. (2014).

CMIST scores for *C. fragile:*

Mean adjusted Likelihood of Invasion: 2.40

Mean adjusted Impact of Invasion: 2.42

Mean adjusted Overall CMIST score: 5.83

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Codium fragile* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports found from Hudson Bay (Stewart and Lockhart, 2005, Mathieson et al., 2010, Küpper et al., 2016).

Q2- Rate of introduction (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Codium fragile* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread in connected ecoregions and could be transported in ballast or through biofouling, but no studies have actually found it in vessels arriving to the Arctic and it is not established in nearby ecoregions, so natural spread is unlikely.

Background Information: *Codium fragile* is present in ports that are directly connected by shipping to Churchill, Deception Bay, Iqaluit, Kuujjuaq, Baker Lake and Chesterfield Inlet and could potentially be entrained and transport by these vessels in hull fouling (Chan et al., 2012) or ballast (Drouin and McKindsey, 2007; and references therein).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Codium fragile.* There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: *Codium fragile* is found on hard bottom surfaces and rocky shores (Fact sheet invasive species from Alaska) and is common in estuarine and shallow embayments (Mathieson et al., 2008b). It is common on the open coast; growing within tide pools in the middle intertidal zone at exposed habitats; also on pilings and rocks in the low intertidal/shallow subtidal of protected sites (Mathieson et al., 2008b). It attaches to rocks, shells, or other hard substrates (Villalard-Bohnsack, 1995). It also can be found in areas with extremely oligotrophic to eutrophic nutrients (Chapman, 1998). In its native range, it has the capability of taking advantage of disturbed habitats, which may help to explain why it has been such a successful invader in other regions of the world (Chavanich et al., 2006). Asynchrony in growing seasons of kelp and *Codium* may favor the establishment and spread of the invasive alga during periods when kelps are most vulnerable (due to herbivory and invasion of the bryozoan *Membranipora membranacea*) (Scheibling and Gagnon, 2006; and references therein). Heavy ice conditions, low salinity, and low air and water temperatures are the most plausible factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Codium fragile*. It can tolerate minimum temperatures of -2°C and it can survive severe winter conditions and initiate new growth in the spring. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: *Codium fragile* tolerates warm average temperatures of >10°C and cold average temperatures of >0°C. Its optimum salinity ranges from 30-35 PSU (https://www.cabi.org/isc/datasheet/107769). It can tolerate large variations in salinity and temperature, enabling it to colonise a wide range of environments (http://www.iucngisd.org/gisd/speciesname/Codium+fragile+ssp.+tomentosoides). In east Asia (Japan and Korea), where *C. fragile* is a native species, its spread is restricted to areas where water temperatures are between 10-20 °C (Lee and Kang, 1986, Segawa, 1996). However, when invading new habitats, it can withstand temperatures as low as -2 °C (Fralick and Mathieson, 1972). A minimal temperature of 10 to 13 °C is required for growth under field conditions (Hanisak, 1979; and references therein). Although growth and reproduction are confined to warmer months, this species can survive severe winter conditions and initiate new growth in the spring (Hanisak, 1979). Its ecological success is partially due to its ability to readily obtain nitrogen to meet its nutrition requirements. It can obtain nitrogen from nitrites/nitrates, ammonia, urea and to a lesser extent from nitrogen fixing bacteria (Prince William Sound Regional Citizens' Advisory Council, 2004). The nitrogen-fixing bacterium that appears to be specifically symbiotic with it may be important to the system (Hanisak, 1979; and references therein). Nutrient supply seems to be important for the establishment and production of the benthic algal communities. It is possible that the absence of an algal community in an otherwise suitable habitat could be directly related to an inadequate supply of nutrients (Dunbar, 1968). Heavy ice conditions, low salinity, and low air and water temperatures are the most plausible factors to consider in attempting to determine the limitations of intertidal benthic algaes (Lee, 1973). *Codium* tolerates salinities from 12 - 17.5 to 40 - 42 ‰ and temperatures from -2 to 27 - 33˚C and thus may colonize both estuarine and marine habitats (Hanisak, 1979, Carlton and Scanlon, 1985, Trowbridge, 1999). This tolerance to a wide range of environmental conditions is reflected in its invaded range throughout the world: from northern Africa to Norway and throughout the Mediterranean in the eastern Atlantic, in the NW Atlantic, on the western coast of the United States, in Chile, Argentina and South Africa as well as New Zealand and Australia (Trowbridge, 1999, Harris and Jones, 2005, Provan et al., 2005).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are sometimes available for *Codium fragile* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature between 12-15°C resulted in selecting a small region of the RA area, but this area could be increased due to its great ability to reproduce asexually. The species requirements are well demonstrated and there are reliable environmental data layers for the RA to show a match.

Background Information: *Codium fragile* needs temperatures between 12-15°C, for at least part of the year for successful sexual reproduction. It has the capacity to spread rapidly via asexual reproduction and fragmentation through fragmentation of the thallus (https://www.cabi.org/isc/datasheet/107769) or sexual, parthenogenetic and vegetative reproduction (https://www.cabi.org/isc/datasheet/107769 and references therein). Growth and reproduction - both in laboratory cultures and in the field - are still possible at approximately 12°C (Chapman, 1998; and references therein). *Codium* grows most rapidly during the summer and fall, and undergoes extensive fragmentation during winter as a means of asexual propagation (Scheibling and Gagnon, 2006; and references therein). Increased wave action during winter storms also reduces *Codium* cover by detachment of whole plants, although remnants of the basal holdfast can regenerate a new thallus the following spring (Scheibling and Gagnon, 2006; and references therein). Hanisak (1979) suggests that reproductive cells may germinate when the salinity is greater than 18 ‰ and the temperature is greater than 12°C.

Q6- Establishment (natural control agents) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents are unlikely to affect population growth in the RA area for *Codium fragile.* Although some taxa in the RA area could act as grazers, there is substantial evidence that this species of algae is generally not preferred over native algal species and grazers are not able to control growth rates. There is considerable published information on the species and its interactions, and there are clear mechanisms for prevention of grazing.

Background Information: Natural enemies of *C. fragile* are usually herbivores such as molluscs, crustaceans, and echinoderms. However, because these herbivores are small and *C. fragile* is not a palatable alga, the impact of herbivores on the algae populations is quite low. One group of snails of the genus *Sacoglossa*, typically found associated with siphonaceous algae, contain several species that are known to feed on *C. fragile* (https://www.cabi.org/isc/datasheet/107769 and references therein). *Codium* is consumed worldwide by a variety of urchins and other herbivores (Prince and LeBlanc, 1992, Trowbridge, 1995).

From Drouin and McKindsey (2007): "Native grazers feed less on *Codium* than on native alga. Chavanich and Harris (2004), observed that the abundance of the snail *Lacuna vincta* decreased in habitat dominated by *Codium* and that individuals that fed on the invasive alga were smaller. *Codium* tissues contain chemical compounds that may act as a protection from certain grazers. Experiments on the feeding behaviour of sea urchins have demonstrated that although *Codium* chemical defenses did not inhibit grazing by sea urchins, they did limit how often *Codium* was grazed upon (Lyons and Scheibling, 2007). What's more, the growth of sea urchins was reduced when feeding solely on *Codium*. Thus, when sea urchins may choose, they avoid feeding on *Codium*, but they can eat it if they have no other choices. Although few species graze on *Codium*, some, such as sea slugs (*Placida dendritica* and *Elysia maoria*), may do so (Freeman and Smith, 1999, Trowbridge and Todd, 2001, Trowbridge, 2002, Bégin and Scheibling, 2003) and may even be responsible for a local decrease in *Codium* populations when occurring at unusually high densities (Trowbridge, 2002, Harris and Jones, 2005). That being said, none of the published literature has demonstrated that herbivory can act as a significant factor in the control of *Codium* invasion. Changes in (for example) grazer populations related to the establishment of *Codium* may also affect higher trophic levels but no data is available to quantify this."

Grazing had no effect on the cover of the corticated green alga *Codium fragile* ssp. *tomentosoides* (currently accepted name *Codium fragile* ssp. *fragile*) in a 13-week field experiment, while the native competitor *Laminaria longicruris* (currently accepted name *Saccharina latissima*) could not persist and cover of turf algae strongly decreased under natural sea urchin densities. But again, *Codium fragile* ssp. *tomentosoides* had superior competitive abilities compared to *Laminaria longicruris* even in the absence of herbivores (Sumi and Scheibling, 2005).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* has the capacity for a wide range of natural dispersal in the RA area. Asexual reproduction is very likely to help the natural spread in the assessment area. There is good evidence for widespread natural dispersal.

Codium fragile has the capacity to spread rapidly via asexual reproduction and fragmentation (https://www.cabi.org/isc/datasheet/107769). It can spread by water currents (Carlton and Scanlon, 1985). *Codium* grows most rapidly during the summer and fall, and undergoes extensive fragmentation during winter by means of asexual propagation (Scheibling and Gagnon, 2006; and references therein). In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* has the capacity for a wide range of anthropogenic dispersal in the RA area. Algae could possibly be taken up in ballast, and is known to foul ships or boats and associated structures (anchors, etc.) and become entangled in fishing implements (e.g. nets, scallop dredges).

Background Information: There are three potential vectors for the transoceanic and/or transcontinental dispersal of *C. fragile*: 1) fouling of the flat oyster *Ostrea edulis* from Europe, 2) fouling of the Japanese or Pacific oyster *Crassostrea gigas* from Washington and British Columbia, 3) fouling of ship's hulls from Europe (Carlton and Scanlon, 1985). It also can be spread via boating, packing material for fishery products (such as lobsters and bait worms). Chan et al. (2012) highlight this species as being present in ports that are connected to Canadian Arctic ports (Churchill and Deception Bay) through hull fouling. Hull fouling transported by domestic coastal resupply and other boat traffic, could spread *C. fragile* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012).

From Hewitt et al. (2007): "Observations and anecdotal evidence indicate that introduced species often become entangled in fishing gear such as nets and ropes, anchor ropes and chains (e.g., Carlton and Scanlon, 1985, Trowbridge, 1995, Trowbridge, 1996, 1998, Relini et al., 2000), possibly leading to further spread of these species. Macroalgal species tolerant to emersion could be successfully transported by these vectors. For example, *Caulerpa taxifolia* and *Codium fragile* ssp. *tomentosoides* survive emersion in high humidity for up to 10 and 90 days, respectively (Sant et al., 1996, Schaffelke and Deane, 2005)."

With the exception of ship-related movements, the importance of other types of movements (fishing and personal vessels) in HBC is not well known.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* would have high impacts in many areas on population growth in the RA area. Much is known of the species impacts on various spatial and temporal scales, both through observational and manipulative studies.

Background Information: *Codium fragile* successfully recruits to gaps in kelp beds and, once established, inhibits recruitment of kelp; and with the presence of another invader (*Membranipora*), it reduces growth, abundance and survival of kelp, resulting in defoliation of kelp plants and displacing by gap formation in kelp beds (Harris and Tyrrell, 2001, Levin et al., 2002). *Codium fragile* exhibits competitive advantages over native seaweeds through opportunistic exploitation of disturbance generated gaps within kelp beds. Once established as dense meadows, *C. fragile* can prevent re-colonization by kelp and persist as the dominant canopy-forming seaweed for prolonged periods (Scheibling and Gagnon, 2006).

Invasion by *Codium* could potentially lead to the local elimination of eelgrass beds (Drouin et al., 2012) as it impacted the abundance and density of eelgrass in eelgrass beds (Drouin and

McKindsey, 2007). It can negatively affect seagrass survival (Garbary et al., 2004) and growth by shading under the algal cover (Drouin et al., 2012). In James Bay, eelgrass beds offer habitat for many marine species and a food source for large populations of migratory waterfowl (Stewart and Lockhart, 2005). Drouin et al. (2011) did find that *Codium* increased fish abundances (*Apeltes quadracus* and *Tautogolabrus adspersus*) in eelgrass beds and invertebrate diversity metrics. From Drouin and McKindsey (2007): "The establishment of *Codium* may affect various habitat characteristics. For example, water circulation may be decreased and sedimentation rates increased as *Codium* creates more dense beds than do native algae (Chapman, 1998). Consequently, shade and turbidity may affect benthic faunal communities and reduce the establishment of native seaweeds. *Codium* morphology may also alter benthic habitats as its bushy thallus differs markedly from kelp blades and eelgrass. Levin et al. (2002) suggest that recruitment of cunner (*Tautogalabrus adspersus*) is greater in native kelp beds than in *Codium* beds. The presence of *Codium* may also alter the recruitment of benthic invertebrates (e.g. barnacles, tunicates, bryozoans), since it may be toxic for some larvae (C.D. Trowbridge, Oregon State University, Newport, United States, personal communication). A reduction of small and sedentary species has been observed under *Codium* canopies, suggesting that this invasive species may contribute to biodiversity loss (Scheibling, 2001). In contrast, Harris and Jones (2005) propose that *Codium* communities might be more diverse and complex than native kelp beds. Bulleri et al. (2006) showed that the presence of *Codium* may enhance the recruitment of mussels in Italy. Likewise, there are also some suggestions that *Codium* may provide shelter to a variety of small fish and invertebrates from predators (A. Drouin, personal observation).

In Nova Scotia, *Codium* establishment has changed the ecological cycling between sea urchin barrens and kelp forests described by Scheibling (1986). Historically the presence of dense kelp populations increases the abundance of sea urchins (*Strongylocentrotus droebachiensis*). Excessive grazing by urchins creates barrens and the lack of food and/or disease increases their mortality, allowing the kelp to dominate once more. The temporary absence of kelp during this cycle now allows for the establishment of *Codium*, which reduces kelp recruitment and gradually replaces this community (Elner and Vadas Sr, 1990, Chapman, 1998, Scheibling et al., 1999, Scheibling, 2001, Chapman et al., 2002). The appearance of *Codium* in this natural cycle has led to changes in species composition (Harris and Tyrrell, 2001) and ten years after the start of the invasion, *Codium* has replaced kelp in some habitats along 900 km of coastline (Scheibling, 2001). The arrival of the invasive bryozoans *Membranipora membranacea* may have facilitated the establishment of *Codium* by increasing kelp mortality (Scheibling et al., 1999, Levin et al., 2002). More recently, *Codium* has also been observed to affect eelgrass habitat. *Codium* may develop filaments that permit it to attach to and grow on *Z. marina* rhizomes (Garbary et al., 2004). Laboratory experiments have shown that rhizomatous growth by *Codium* induced 90% mortality of *Z. marina* in four months (Garbary et al., 2004). This particular type of growth may be an adaptation to environmental pressure when developing on soft bottom habitat.

Altered algal community composition may also modify various biotic interactions. Some authors have highlighted *Codium*’s affinity for nitrogen, which might result in competition between this species, phytoplankton and other macroalgae (Hanisak and Harlin, 1978, Chapman, 1998). Native grazers feed less on *Codium* than on native algae. Chavanich and Harris (2004) observed that the abundance of the snail *Lacuna vincta* decreased in habitat dominated by *Codium* and that individuals that fed on the invasive alga were smaller. *Codium* tissues contain chemical compounds that may act as a protection from certain grazers. Experiments on the feeding behaviour of sea urchins have demonstrated that although *Codium* chemical defences did not inhibit grazing by sea urchins, they did limit how often *Codium* was grazed upon (Lyons et al., 2006). What’s more, the growth of sea urchins was reduced when feeding solely on *Codium*. Thus, when sea urchins may choose, they avoid feeding on *Codium*, but they can eat it if they have no choice. Although few species graze on *Codium*, some, such as sea slugs (*Placida dendritica* and *Elysia maoria*), may do so (Freeman and Smith, 1999, Trowbridge and Todd, 2001, Trowbridge, 2002, Bégin and Scheibling, 2003) and may even be responsible for a local decrease in *Codium* populations when occurring at unusually high densities (Trowbridge, 2002, Harris and Jones, 2005). That being said, none of the published literature has demonstrated that herbivory can act as a significant factor in the control of *Codium* invasion. Changes in grazer populations related to the establishment of *Codium* may also affect higher trophic levels but no data is available to quantify this."

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* would have high impacts in many areas on communities in the RA area. Much is known of the species impacts on various spatial and temporal scales, both through observational studies and manipulative studies.

Background Information: It can affect community structure and composition affecting other native species by shifting habitat selection and feeding behaviours (Chavanich and Harris, 2004, Harris and Jones, 2005). It can decrease the epifauna diversity and density, and increase the epiflora density (Schmidt and Scheibling, 2006).

From Drouin and McKindsey (2007): "The establishment of *Codium* may affect various habitat characteristics. For example, water circulation may be decreased and sedimentation rates increased as *Codium* creates more dense beds than do native algae (Chapman, 1998). Consequently, shade and turbidity may affect benthic faunal communities and reduce the establishment of native seaweeds. *Codium* morphology may also alter benthic habitats as its bushy thallus differs markedly from kelp blades and eelgrass. Levin et al. (2002) suggest that recruitment of cunner (*Tautogalabrus adspersus*) is greater in native kelp beds than in *Codium* beds. The presence of *Codium* may also alter the recruitment of benthic invertebrates (e.g. barnacles, tunicates, bryozoans), since it may be toxic for some larvae (C.D. Trowbridge, Oregon State University, Newport, United States, personal communication). A reduction of small and sedentary species has been observed under *Codium* canopies, suggesting that this invasive species may contribute to biodiversity loss (Scheibling, 2001). In contrast, Harris and Jones (2005) propose that *Codium* communities might be more diverse and complex than native kelp beds. Bulleri et al. (2006) showed that the presence of *Codium* may enhance the recruitment of mussels in Italy. Likewise, there are also some suggestions that *Codium* may provide shelter to a variety of small fish and invertebrates from predators (A. Drouin, personal obsservation).

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Codium impacted the abundance and density of eelgrass in eelgrass beds (Drouin and Mckindsey, 2007) as well as increased the abundance of two species of fish and invertebrate diversity (Drouin et al. 2011).

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* would have high impact in many areas on habitat in the RA area. Much is known of the species impacts on various spatial and temporal scales, both through observational and manipulative studies.

Background Information: In general, these types of algae can form mats or uniform meadows, which also may change the existing architectural structure from a complex three-dimensional system, formed by large seaweeds and phanerogams, into an almost two-dimensional one, with implications for other species in the community and their contributions to ecosystems (Wallentinus and Nyberg, 2007). As *C. fragilis* is an opportunistic species, its spread can alter the ecosystem by replacing the dominant species such as kelp (Harris and Tyrrell, 2001). The morphological structure of *C. fragile* ssp. *tomentosoides* will likely increase sedimentation since it is a "low lying" alga, making it difficult for some large invertebrates and fish to move among the plants and live in the space between the bushy parts of the algae and the seabed (the understory) (http://www.iucngisd.org/gisd/speciesname/Codium+fragile+ssp.+tomentosoides)

From Drouin and McKindsey (2007): "The establishment of *Codium* may affect various habitat characteristics. For example, water circulation may be decreased and sedimentation rates increased as *Codium* creates more dense beds than do native algae (Chapman, 1998). Consequently, shade and turbidity may affect benthic faunal communities and reduce the establishment of native seaweeds. *Codium* morphology may also alter benthic habitats as its bushy thallus differs markedly from kelp blades and eelgrass. In Nova Scotia, *Codium* establishment has changed the ecological cycling between sea urchin barrens and kelp forests described by Scheibling (1986). Historically the presence of dense kelp populations increases the abundance of sea urchins (*Strongylocentrotus droebachiensis*). Excessive grazing by urchins creates barrens and the lack of food and/or disease increases their mortality, allowing the kelp to dominate once more. The temporary absence of kelp during this cycle now allows for the establishment of *Codium*, which reduces kelp recruitment and gradually replaces this community (Elner and Vadas Sr, 1990, Chapman, 1998, Scheibling et al., 1999, Scheibling, 2001, Chapman et al., 2002). The appearance of *Codium* in this natural cycle has led to changes in species composition (Harris and Tyrrell, 2001) and ten years after the start of the invasion, *Codium* has replaced kelp in some habitats along 900 km of coastline (Scheibling, 2001). The arrival of the invasive bryozoans *Membranipora membranacea* may have facilitated the establishment of *Codium* by increasing kelp mortality (Scheibling et al., 1999, Levin et al., 2002). More recently, *Codium* has also been observed to affect eelgrass habitat. *Codium* may develop filaments that permit it to attach to and grow on *Z. marina* rhizomes (Garbary et al., 2004). Laboratory experiments have shown that rhizomatous growth by *Codium* induced 90% mortality of *Z. marina* in four months (Garbary et al., 2004). This particular type of growth may be an adaptation to environmental pressure when developing on soft bottom habitat.

Altered algal community composition may also modify various biotic interactions. Some authors have highlighted *Codium's affinity* for nitrogen, which might result in competition between this species, phytoplankton and other macroalgae (Hanisak and Harlin, 1978, Chapman, 1998)."

Codium impacted the abundance and density of eelgrass in eelgrass beds (Drouin and Mckindsey, 2007) as well as increased the abundance of two species of fish and invertebrate diversity (Drouin et al. 2011).

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* would have high impact in many areas on ecosystem function in the RA area. Much is known of the species impacts on various spatial and temporal scales, both through observational studies and manipulative studies.

Background Information: It can affect community structure and composition affecting other native species by shifting habitat selection and feeding behaviors (Chavanich and Harris, 2004, Harris and Jones, 2005). The morphological structure of *C. fragile* ssp. *tomentosoides* will likely increase sedimentation since it is a "low lying" alga, making it difficult for some large invertebrates and fish to move among the plants and live in the space between the bushy parts of the algae and the seabed (the understory) (http://www.iucngisd.org/gisd/speciesname/Codium+fragile+ssp.+tomentosoides).

From Drouin and McKindsey (2007): "The establishment of *Codium* may affect various habitat characteristics. For example, water circulation may be decreased and sedimentation rates increased as *Codium* creates more dense beds than do native algae (Chapman, 1998). Consequently, shade and turbidity may affect benthic faunal communities and reduce the establishment of native seaweeds. *Codium* morphology may also alter benthic habitats as its bushy thallus differs markedly from kelp blades and eelgrass. Levin et al. (2002) suggest that recruitment of cunner (*Tautogalabrus adspersus*) is greater in native kelp beds than in *Codium* beds. The presence of *Codium* may also alter the recruitment of benthic invertebrates (e.g. barnacles, tunicates, bryozoans), since it may be toxic for some larvae (C.D. Trowbridge, Oregon State University, Newport, United States, pers. com.). A reduction of small and sedentary species has been observed under *Codium* canopies, suggesting that this invasive species may contribute to biodiversity loss (Scheibling, 2001). In contrast, Harris and Jones (2005) propose that *Codium* communities might be more diverse and complex than native kelp beds. Bulleri et al. (2006) showed that the presence of *Codium* may enhance the recruitment of mussels in Italy. Likewise, there are also some suggestions that *Codium* may provide shelter to a variety of small fish and invertebrates from predators (A. Drouin, pers. obs.).

In Nova Scotia, *Codium* establishment has changed the ecological cycling between sea urchin barrens and kelp forests described by Scheibling (1986). Historically the presence of dense kelp populations increases the abundance of sea urchins (*Strongylocentrotus droebachiensis*). Excessive grazing by urchins creates barrens and the lack of food and/or disease increases their mortality, allowing the kelp to dominate once more. The temporary absence of kelp during this cycle now allows for the establishment of *Codium*, which reduces kelp recruitment and gradually replaces this community (Elner and Vadas Sr, 1990, Chapman, 1998, Scheibling et al., 1999, Scheibling, 2001, Chapman et al., 2002). The appearance of *Codium* in this natural cycle has led to changes in species composition (Harris and Tyrrell, 2001) and ten years after the start of the invasion, *Codium* has replaced kelp in some habitats along 900 km of coastline (Scheibling, 2001). The arrival of the invasive bryozoans *Membranipora membranacea* may have facilitated the establishment of *Codium* by increasing kelp mortality (Scheibling et al., 1999, Levin et al., 2002). More recently, *Codium* has also been observed to affect eelgrass habitat. *Codium* may develop filaments that permit it to attach to and grow on *Z. marina* rhizomes (Garbary et al., 2004). Laboratory experiments have shown that rhizomatous growth by *Codium* induced 90% mortality of *Z. marina* in four months (Garbary et al., 2004). This particular type of growth may be an adaptation to environmental pressure when developing on soft bottom habitat.

Altered algal community composition may also modify various biotic interactions. Some authors have highlighted *Codium's* affinity for nitrogen, which might result in competition between this species, phytoplankton and other macroalgae (Hanisak and Harlin, 1978, Chapman, 1998). Native grazers feed less on *Codium* than on native algae. Chavanich and Harris (2004) observed that the abundance of the snail *Lacuna vincta* decreased in habitat dominated by *Codium* and that individuals that fed on the invasive alga were smaller. *Codium* tissues contain chemical compounds that may act as a protection from certain grazers. Experiments on the feeding behaviour of sea urchins have demonstrated that although *Codium* chemical defences did not inhibit grazing by sea urchins, they did limit how often *Codium* was grazed upon (Lyons et al., 2006). What's more, the growth of sea urchins was reduced when feeding solely on *Codium*. Thus, when sea urchins may choose, they avoid feeding on *Codium*, but they can eat it if they have no choice. Although few species graze on *Codium*, some, such as sea slugs (*Placida dendritica* and *Elysia maoria*), may do so (Freeman and Smith, 1999, Trowbridge and Todd, 2001, Trowbridge, 2002, Bégin and Scheibling, 2003) and may even be responsible for a local decrease in *Codium* populations when occurring at unusually high densities (Trowbridge, 2002, Harris and Jones, 2005). That being said, none of the published literature has demonstrated that herbivory can act as a significant factor in the control of *Codium* invasion. Changes in (for example) grazer populations related to the establishment of *Codium* may also affect higher trophic levels but no data is available to quantify this."

Codium impacted the abundance and density of eelgrass in eelgrass beds (Drouin and Mckindsey, 2007) as well as increased the abundance of two species of fish and invertebrate diversity (Drouin et al. 2011).

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Codium fragile* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers/other species in the RA area. There are a wide variety of species that could be transported, but few appear to have major impacts with the exception of *Allobodo chlorophagus* n. gen., n. sp. which appears to be highly host-specific to *C. fragile*. There is relatively limited general knowledge of effects of diseases and epiphytes on this algae.

Background Information: *Codium fragile* is often covered with epiphytic species (Villalard-Bohnsack, 1995). The red algae *Ceramium nodulosum* is a commonly found epiphyte, together with at least 17 species of other epiphytic algae and several groups of herbivores such as molluscs, crustaceans, echinoderms, and bryozoans that use *Codium fragile* as their habitat and food source (https://www.cabi.org/isc/datasheet/107769 and references therein). *Codium fragile* can host an epiphytic algae, *Neosiphonia harveyi*, which is also a NIS from the Pacific that has invaded the Northwest Atlantic (Mathieson et al., 2008a). This algae can grow abundantly on *Codium*, which enhances its development and transport via drifting fragments of *Codium* (Bartsch and Kuhlenkamp, 2000).

From Egan et al. (2014): "Pathogenic pressure by bacteria might be substantial, as macroalgae have evolved a range of innate and induced defense mechanisms that have the potential to control bacterial attacks. The presence and abundance of virulence factors in marine bacteria, which have not previously been recognized as pathogens, also represents an underappreciated, opportunistic potential for disease. Given that virulence expression in opportunistic pathogens is often dependent on environmental conditions, we predict that current and future anthropogenic changes in the marine environment will lead to an increase in the occurrence of macroalgal disease."

From Kerrison et al. (2015): "Little is currently known about diseases in kelp, or seaweeds in general, although various causative agents have been implicated (Gachon et al., 2010). Problematic diseases/syndromes have been identified in large scale cultivation of kelp over the last half century (Scoggan et al., 1989) and their occurrence has increased as cultivation has intensified, with a 4-5% yield loss now reported in Korean cultivation (Gachon et al., 2010). The bacteria *Pseudoalterom* spp. and *Alteromonas* spp. are known to be responsible for some disease (Egan et al., 2014), but in numerous cases, the agent has not been identified. The prevalence of endophytic infection is known to be high in wild kelp populations (Ellertsdottir and Peters, 1997), and so there are concerns that a) pathogens may be transplanted with seaweed stocks, infecting nearby natural seaweed beds and b) as physicochemical stress is often a trigger for outbreaks in cultivated kelp (Scoggan et al., 1989), climate change impacts such as rising seawater temperatures may in the future lead to more severe disease impacts."

From Goodwin et al. (2018): "[*Allobodo chlorophagus* n. gen., n. sp.], a novel biflagellate protist that consumed chloroplasts inside material of the invasive marine green alga *Codium fragile* was reported from the U.S. east coast in 2003. We observed a similar association in *C. fragile* from five sites in Nova Scotia, Canada during 2013 and 2014. The lack of evidence for food sources other than *Codium* is consistent with a parasitic association, but other possibilities exist (e.g. necrotrophy)."

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: There is no information on other *Codium* species present in the assessment area (Mathieson et al., 2010) (https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Codium fragile* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some benthic species at risk that could be impacted given the general demonstrated effects of invasive algae on benthic species. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through habitat, food web and ecosystem impacts - these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; thorny skate, *Amblyraja radiata*. Walrus populations of risk/special concern may also be impacted as they feed on benthic invertebrates so may be impacted through changes in food resources in benthic habitats by this algae.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Codium fragile* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. There are possible slight impacts to fisheries species. Aquaculture is not really practiced in the area.

Background Information: *Codium fragile* can overgrow mussel beds and oyster reefs, directly affecting economic gains. It can even float away with oysters and mussels (Wallentinus and Nyberg, 2007; and references therein). *Codium fragile* ssp. *tomentosoides* can have economic impacts on shellfish and fishing industries. It can smother mussels and scallops, reduce biomass of oysters, lift shellfish off the seafloor, foul nets, and clog scallop dredges (Trowbridge, 1999). This leads to an increase in labor costs during harvesting and processing, associated with removal of the algae (Carlton and Scanlon, 1985, Trowbridge, 1999). It can reduce productivity and coverage of eelgrass beds (Wong and Vercaemer, 2012). In James Bay, these plants offer habitat for many marine species and a food source for large populations of migratory waterfowl that are hunted locally for subsistence and sport (Stewart and Lockhart, 2005). Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Codium fragile* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: *Codium fragile* has spread throughout the northern and southern hemispheres including the northeastern and northwestern Atlantic, the Mediterranean, Australia, New Zealand, the eastern central Pacific, and southeastern Pacific. There have been broad geographic expansions, extending from the Gulf of St Lawrence to Newfoundland. The initial population of *Codium* at the Isles of Shoals (1983) (Northwest Atlantic) increased to 27 sites in 22 years (Mathieson et al., 2008a). It has been ranked as one of five top risk species among marine macroalgae in Europe by Nyberg and Wallentinus (2005).

Dumontia contorta (S.G.Gmelin) Ruprecht, 1850

Phylum: Rhodophyta

Class: Florideophyceae

Order: Gigartinales

Family: Dumontiaceae

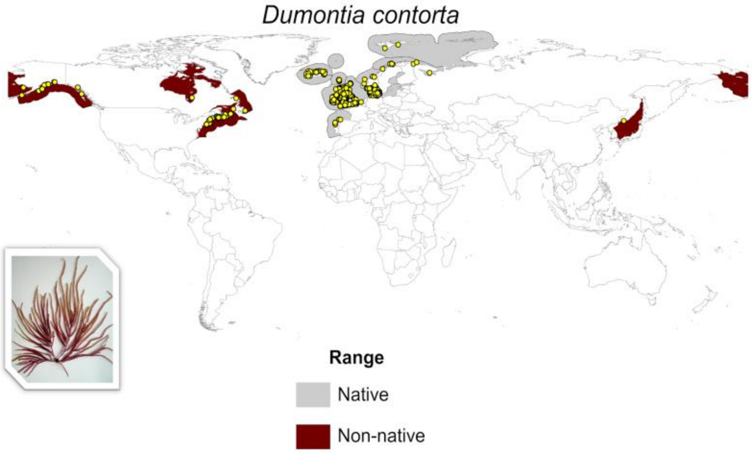

Figure 24: Ecoregions where Dumontia contorta is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Dumontia contorta occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (https://doi.org/10.15468/dl.rh4ycf, 23 November 2017), Mathieson et al. (2008b), Kozhenkova (2009), G. Moore pers comm 2018. Picture of D. contorta modified from https://www.flickr.com/photos/seaweedlady/2506547800.

CMIST scores for *D. contorta:*

Mean adjusted Likelihood of Invasion: 2.56

Mean adjusted Impact of Invasion: 1.95

Mean adjusted Overall CMIST score: 4.98

Q1- Present status in the area (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Dumontia contorta* is established in the RA area. The species has been identified on more than one occasion in a small region of the assessment area and has also been reported elsewhere in the Canadian Arctic, suggesting it is established. Note that survey effort is low relative to the size of the RA area.

Background Information: Howe (1927) reported *D. incrassata*, a synonym of *D. contorta*, on mud bottom in 3-5 m depth at Goose Island in southern James Bay, where it was described as rare. Mathieson et al. (2010) described it for James Bay. Lee (1980) found a few small-sized thalli of *D. incrassata* in the loose-lying community at 3-8 m depth on the mud bottom of Sachs Harbour, Banks Island. Bell and MacFarlane (1938) reported it from Dundas Harbour and Pond Inlet as *D. filiformis*, another synonym. Specimens identified as *Dumontia contorta* occur in both the North Atlantic and the North Pacific oceans but are absent from the Arctic Ocean (Lindstrom, 2001). These Atlantic and Pacific algae are thought to represent distinct species (*D. contorta* in the Atlantic and *D. alaskana* in the Pacific; Tai et al., 2001).

Q2- Rate of introduction (Score = 3, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* could arrive frequently in high numbers in the RA area. It is widespread in connected ecoregions and could be transported in ballast or through biofouling (commercial or fishing vessels) and is established in adjacent ecoregions (Labrador). Although no studies have actually found it in vessels arriving in the Arctic it has been recorded in the RA and elsewhere.

Background Information: There is no specific information for this species. Looking at its distribution (Q17), ships from Europe and some from the Northwest Atlantic could act as vectors of introduction through ballast and biofouling in the assessment region, but there are no values for frequency and propagule pressure.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Dumontia contorta.* There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: *Dumontia contorta* is common and widely distributed on rock and pebbles, shallow open pools, and in the upper intertidal to shallow subtidal zone (https://www.seaweed.ie/descriptions/Dumontia_contorta.php). It grows in abundance in tufts in small tide pools and also on rocks that are exposed to the air at low water (Dunn, 1917). The absence of distinct latitudinal ecotypes in *D. contorta* permits one practical conclusion: it is possible to use the temperature day length responses of one strain to infer the approximate geographic boundaries for this species (Rietema and Van den Hoek, 1984). Heavy ice conditions, low salinity, and low air and water temperatures are the most plausible factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Dumontia contorta.* Even though it has been studied that minimum temperature tolerance is between 0° and -2° C, it can be affected by freezing air temperatures when they are exposed. Species already occurs in the RA area and it has been found elsewhere in the Arctic. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Temperatures as low as 0°C can be survived (Pot et al., 1988) and it is even capable of surviving winter temperatures as low as 0 to -2°C, probably even lower (Rietema and Van den Hoek, 1984). Nutrient supply seems to be important for the establishment and production of the benthic algal communities. It is even possible that the absence of an algal community in an otherwise suitable habitat could be directly related to an inadequate supply of nutrients (Dunbar, 1968). Being in rock pools makes it more susceptible to freezing damage. Five species of brown algae, including *Dumontia incrassata*, showed 70 and 80 % of tissue water frozen after 3 h at -20°C and reduction of 67% of photosynthesis after a single 12 h freezing event (Davison et al., 1989). Heavy ice conditions, low salinity, and low air and water temperatures are the most plausible factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q5- Establishment (reproductive requirements) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are almost always available for *Dumontia contorta* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Mean sea surface and bottom temperature >=4°C (temperature required to start reproduction) together with the required hours of daylight, which resulted in selecting 2/3 of the RA coastline area. The species requirements for reproduction are well demonstrated and there are reliable environmental data layers for the RA area to show a match of about 2/3 of the area; species also known to occur within the RA area and other areas of the Canadian Arctic.

Background Information: This species must develop erect/upright fronds from the crust to reproduce (Van den Hoek et al., 1990). Upright fronds from the crusts have been shown to be affected by short-day length with a critical day length of about 12-13 h and can take place in a broad temperature range of about 4-18°C (Pot et al., 1988). In the Wadden Sea, it flourishes in the winter. That is, it germinates in the autumn on stones and develops long thin fronds in late winter and early spring (Ecomare). *Dumontia contorta* behaves as a winter-spring annual; upright fronds appear in autumn or winter, achieving maximum development during the winter-spring period (Pot et al., 1988). There have been no latitudinal temperature-day length ecotypes found in this species (Rietema and Van den Hoek, 1984). The lowest temperature that inhibits initiation of macrothalli is 0°C , whereas macrothalli are initiated at 4°C, although with much retardation (Rietema and Van den Hoek, 1984). A summer temperature of > 4°C is just high enough for a minimal initiation of macrothalli (Rietema and Van den Hoek, 1984). Vegetative reproduction is quite common in red algae. Thallus fragmentation is considered by many as the most significant kind of vegetative reproduction in red algae (Gurgel and Lopez-Bautista, 2007) and this family is known for its ability to fragment (Williams and Smith, 2007).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Dumontia contorta.* There are closely related species in the RA area that could act as predators. There is limited information on predators and the available information is conflicting.

Background Information: The herbivorous marine snail *Littorina littorea* controls the abundance and type of algae in high intertidal tide pools in New England (Lubchenco, 1978). Periwinkle has a medium food preference for *Dumontia* spp. (Lubchenco, 1978). This grazing could be done by the native congeners *L. obtusata* and *L. saxatilis* in the assessment region. That said, a study evaluating the life history of *Dumontia contorta* (Pot et al., 1988) state that "the dense populations of *L. littorea* in Lake Grevelingen were not observed to graze on *Dumontia* fronds" and that "Other invertebrates using the fronds for the construction of dwellings were much more destructive."

A review by Enge et al. (2017) found that several non-native red seaweeds were of low palatability to native herbivores.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Dumontia contorta* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for widespread natural dispersal in red algae and macro algal species generally. This information is based on other closely related species (e.g., *Dumontia incrassata*) as there is no information on *D. contorta* itself.

Background Information: Vegetative reproduction is quite common in red algae. Thallus fragmentation is considered by many as the most significant kind of vegetative reproduction in red algae. Branchlets are designed to break apart, disperse and develop into new plants (Gurgel and Lopez-Bautista, 2007). In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Dumontia contorta* has the capacity for a wide range of anthropogenic dispersal in the RA area. Alga may be taken up in ballast, foul ship or boats and associated structures (anchors, etc.) and become entangled in fishing implements (e.g. nets, scallop dredges), however scoring is based on other closely related species (red algae generally) and not evidence for *D. contorta* itself.

Background Information: *Dumontia contorta* is hypothesized to have been introduced from Europe in the Northwest Atlantic in the early 1900s, being initially recorded in Maine (Mathieson et al., 2008b, Mathieson et al., 2010). It is recorded in James Bay and Ellesmere-Baffin Island (Mathieson et al., 2010), although the mode of introduction is unknown (Mathieson et al., 2008b). It could have reached the Arctic through natural range expansion. Lima et al. (2007) documented a northward shift of 62 km (1.3 km/year) in Portugal, which was likely a result of climate change.

From Hewitt et al. (2007): “While the majority of macroalgae associated with hull fouling are small or have crustose or filamentous growth forms, several large species have been collected from hulls.” (*Dumontia contorta* has both crustose and filamentous growth forms).

With the exception of ship-related movements, the importance of other types of movements (fishing and personal vessels) in HBC is not well known.

Vectors for domestic translocation are similar to those of the initial introduction, such as aquaculture stock movements, and coastal and recreational shipping (Schaffelke et al., 2006).

Q9- Impact on population (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* would have high impacts in few areas or moderate impact in many areas on population growth in the RA area. There is no information on potential impacts related to this species, but potential for effects based on it being a habitat-forming species and primary producer. Hence there are many possibilities for impacts to population, based on known general effects of other invasive seaweeds.

Background Information: There is no specific information on impacts of *D. contorta* as an invasive species. The ecological effects of only 17 out of 277 introduced seaweeds have been studied, and red seaweeds have been studied the least (Williams and Smith, 2007). Seaweed impacts in general have shown changes in the abundance of native biota (Williams and Smith, 2007).

Q10- Impact on communities (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There is no information on potential impacts related to this species, but potential for effects based on it being a habitat-forming species and primary producer. Hence, there are many possibilities for impacts to communities based on known general effects of other invasive seaweeds.

Background Information: There is no specific information on impacts of *D. contorta* as an invasive species. The ecological effects of only 17 out of 277 introduced seaweeds have been studied, and red seaweeds have been studied the least (Williams and Smith, 2007). Seaweed impacts in general have shown changes in the abundance of native biota, and some of them could represent significant changes in community structure (Williams and Smith, 2007).

From Williams and Smith (2007): “Community-level ecological interactions involving introduced seaweeds constitute a major research gap. Indirect effects between trophic levels, the mobility of consumers, and restrictions on replication present research challenges. Major shifts in community structure can occur even if species richness and biodiversity remain unchanged (Sax et al., 2005). In addition, resilience of native communities may be reduced after invasion by seaweeds (Valentine and Johnson, 2003, Piazzi and Ceccherelli, 2006).”

Q11- Impact on habitats (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* would have high impacts in few areas or moderate impact in many areas on habitat in the RA area. There is no information on potential impacts related to this species, but potential for effects based on it being a habitat-forming species and primary producer. Hence, there are many possibilities for impacts to habitats based on known general effects of other invasive seaweeds.

Background Information: In general, exotic macroalgae impact native macroalgal community structure and diversity (Nyberg, 2007) by altering the physical, chemical, and biotic characteristics of the habitat. They can act as a substrate of other organisms, especially macrofauna associated with macroalgae in kelp forests such as Bryozoa, Polychaeta and Hydrozoa (Włodarska-Kowalczuk et al., 2009, Ronowicz et al., 2013). They also can act as substrate for other invasive organisms such as *Porphyra* algae (Neefus et al., 2008). *Dumontia contorta* can grow on solid rock and as an abundant epiphyte on *Chondrus crispus* (which is not present in the Canadian Arctic (https://www.gbif.org)) and other mid-low intertidal seaweeds. Therefore, this might represent its major ecological impact locally (Mathieson 2019 personal communication).

Q12- Impact on ecosystem function (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. There is no information on potential impacts related to this species, but potential for effects based on it being a habitat-forming species and primary producer. Hence, there are many possibilities for impacts to ecosystem levels, based on known general effects of other invasive seaweeds.

Background Information: There is no specific information on impacts of *D. contorta* as an invasive species. The ecological effects of only 17 out of 277 introduced seaweeds have been studied, where red seaweeds have been studied the least (Williams and Smith, 2007).

From Williams and Smith (2007) and references therein: “Interactions between introduced seaweeds and native herbivores could alter trophic dynamics and seaweed spread if herbivores avoid eating the invaders. In the majority of feeding experiments, although introduced seaweeds were not preferred by generalist herbivores (littorines, isopods, polychaetes, sea urchins, fishes), they were eaten. Thus, introduced seaweeds do not escape completely from novel herbivores. Despite eating introduced seaweeds, native herbivores have not been documented to control invader spread. These general patterns where herbivores cannot control the spread of introduced seaweeds contrast with a recent meta-analysis showing that native generalist herbivores, particularly large vertebrates, provide biotic resistance to plant invasions on land and in freshwater and saltwater marshes (Parker et al., 2006). These differences may be due to the fact that large vertebrate herbivores are not as common in seaweed-dominated habitats. The hypothesis that introduced seaweeds increase primary productivity, which could lead to higher consumer abundance, requires investigation in natural communities. Generally unpalatable themselves, introduced seaweeds can support high abundances of palatable epiphytes, but epiphyte populations can be notoriously ephemeral food sources for consumers. We found no studies that assessed introduced seaweeds as trophic support for detritivores. Finally, an understanding of how introduced seaweeds alter the flow of matter and energy through ecosystems must be considered along with any effects on biodiversity.”

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Dumontia contorta* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. There are a wide variety of species that could be transported based on general patterns in other seaweeds and some information available specific to *D. contorta*, including two species known to do well in cold regions. There is relatively limited general knowledge of effects of diseases and epiphytes.

Background Information: *Dumontia contorta* can act as a substrate for other invasive organisms such as *Porphyra* algae and *Ulonema rhizophorum* (Mathieson et al., 2008b, Neefus et al., 2008).

From Egan et al. (2014): “pathogenic pressure by bacteria might be substantial, as macroalgae have evolved a range of innate and induced defense mechanisms that have the potential to control bacterial attacks. The presence and abundance of virulence factors in marine bacteria, which have not previously been recognized as pathogens, also represents an underappreciated, opportunistic potential for disease. Given that virulence expression in opportunistic pathogens is often dependent on environmental conditions, we predict that current and future anthropogenic changes in the marine environment will lead to an increase in the occurrence of macroalgal disease.”

From Kerrison et al. (2015): “Little is currently known about diseases in kelp, or seaweeds in general, although various causative agents have been implicated (Gachon et al., 2010). Problematic diseases/syndromes have been identified in large scale cultivation of kelp over the last half century (Scoggan et al., 1989) and their occurrence has increased as cultivation has intensified, with a 4-5% yield loss now reported in Korean cultivation (Gachon et al., 2010). The bacteria *Pseudoalterom* spp. And *Alteromonas* spp. Are known to be responsible for some disease (Egan et al., 2014), but in numerous cases, the agent has not been identified. The prevalence of endophytic infection is known to be high in wild kelp populations (Ellertsdottir and Peters, 1997), and so there are concerns that a) pathogens may be transplanted with seaweed stocks, infecting nearby natural seaweed beds and b) as physicochemical stress is often a trigger for outbreaks in cultivated kelp (Scoggan et al., 1989), climate change impacts such as rising seawater temperatures may in the future lead to more severe disease impacts.”

Q14- Genetic impact **(Score = 1, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Dumontia contorta* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: There is no information on other *Dumontia* species present in the assessment area (Mathieson et al., 2010, https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species **(Score = 2, Certainty = 1)**

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* would have high impacts in few areas or moderate impacts in many areas on at-risk or depleted species in the RA area. There are some benthic species at risk that could be impacted given the demonstrated effects of invasive algae on benthic ecosystems generally. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through direct or indirect habitat, food web and ecosystem impacts – these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and thorny skate, *Amblyraja radiata*. Walrus populations at risk or of special concern may also be impacted as they feed on benthic invertebrates so may be impacted through changes in food resources in benthic habitats by this algae.

Q16- Impact on fisheries **(Score = 2, Certainty = 1)**

Score Rationale: Little to no reliable information is available to suggest *Dumontia contorta* would have high impact in few areas or moderate impact in many areas of fished species in the RA area. There may be slight impacts to fisheries species, similar to impacts caused by other seaweeds, but this is uncertain given lack of information on the species and on red seaweeds generally. Aquaculture is not really practiced in the area.

Background Information: There is no information found, however seaweed impacts in general have shown changes in the abundance of native biota (Williams and Smith, 2007), so impacts to some harvested, particularly benthic species, such as shellfish, are possible. Shellfish are not cultivated in the assessment area, however, natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Dumontia contorta* is invasive elsewhere in the world. There is solid evidence of establishment in invaded areas.

Background Information: From Mathieson et al. (2010): "*Dumontia* is thought to have been introduced [to the northwest Atlantic] from Europe in the early 1900's, being initially recorded in southern Maine (Dunn, 1916, 1917, Mathieson et al., 2008a, Mathieson et al., 2008b) and now known from the northeastern Canadian Arctic to Long Island Sound (Taylor, 1962, Lee, 1980, Sears, 2002). Currently it is one of the most conspicuous red algae in the Northwest Atlantic (Kilar and Mathieson, 1978)."

Sargassum muticum (Yendo) Fensholt, 1955

Phylum: Ochrophyta
Class: Phaeophyceae
Order: Fucales
Family: Sargassaceae

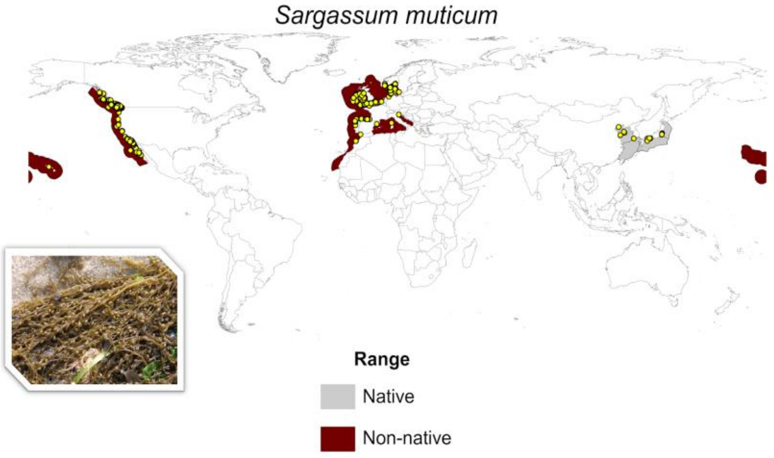

Figure 25: Ecoregions where Sargassum muticum is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution, and occurrence points (in yellow) have not been collected in an exhaustive manner. Sargassum muticum occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.e5wzt2, 4 May 2017), Cheang et al. (2010), Sabour et al. (2013), Sfriso and Facca (2013), El Atouani et al. (2016). Picture of S. muticum modified from https://www.inaturalist.org/guide_taxa/784199.

CMIST scores for *S. muticum:*

Mean adjusted Likelihood of Invasion: 2.40

Mean adjusted Impact of Invasion: 2.42

Mean adjusted Overall CMIST score: 5.82

Q1- Present status in the area **(Score = 1, Certainty = 2)**

Score Rationale: Some reliable information suggests *Sargassum muticum* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports have been found in Hudson Bay (Stewart and Lockhart, 2005, Mathieson et al., 2010, Küpper et al., 2016).

Q2- Rate of introduction **(Score = 2, Certainty = 1)**

Score Rationale: Little to no reliable information is available to suggest *Sargassum muticum* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread in connected ecoregions and could be transported in ballast or through biofouling, although no studies have actually found it in vessels arriving to the Arctic. It is not established in nearby ecoregions, so natural spread is unlikely.

Background Information: *Sargassum muticum* is present in ports that are directly connected by shipping to Churchill and Deception Bay and could potentially be entrained and transported by these vessels in ballast water and by biofouling (Chan et al., 2012).

Q3- Survival (suitable habitat) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Sargassum muticum*. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support the classification.

Background Information: *Sargassum muticum* grows on hard substrates in shallow waters (White, 2010). It forms dense canopies spanning lower intertidal to subtidal regions, most commonly on wave-sheltered shores (Druehl, 2000). Stable boulder substratum (>10 cm in diameter) can facilitate *Sargassum* abundance (Thomsen et al., 2006) and lack of suitable hard substrates may limit its expansion (Stæhr et al., 2000). It can colonize both disturbed habitats (De Wreede, 1983) and more diverse macroalgal assemblages (Sanchez and Fernández, 2005). Eutrophic conditions favour the growth of *S. muticum* (Wallentinus, 2002). Heavy ice conditions, low salinity, and low air and water temperatures are the most plausible factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q4- Survival (suitable environment) **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Sargassum muticum.* The species is known to have survived at -1.4°C, and live at ranges in winter between 1-3°C. Most of the RA area would have suitable salinity. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From CABI: “Where *S. muticum* occurs on the Pacific coast of North America, water temperatures can range between a winter minimum of 1-3ºC in British Columbia to a summer maximum of 18ºC in southern California. Estuarine populations in Oregon tolerate salinities down to 20 ppt. *Sargassum muticum* germling growth increases over the range of 5 to 25ºC and, although capable of growing at 7ºC, growth is much slower than at 17ºC. Ideal conditions for growth are 25ºC and 34 ppt salinity, but with the ability to grow at temperatures of 10 to 30ºC and salinities of 6.4 to 34 ppt. Populations cannot be sustained for prolonged periods below 15 ppt due to the suppression of reproduction.” (https://www.cabi.org/isc/datasheet/108973)

Observations in Swedish waters indicate that *S. muticum* has survived long periods with water temperatures of -1.4°C (Karlsson, 1988) and Lüning and Freshwater, (1988) found survival of *S. muticum* at -1.5ºC for one week. It is highly tolerant to desiccation, full sunlight and variations in salinity and temperature (https://www.seaweed.ie/descriptions/Sargassum_muticum.php). Nutrient supply seems to be important for the establishment and production of benthic algal communities. It is even possible that the absence of an algal community in an otherwise suitable habitat could be directly related to an inadequate supply of nutrients (Dunbar, 1968). Heavy ice conditions, low salinity, and low air and water temperatures are the most important factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are sometimes available for *Sargassum muticum* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: A minimum of 4 months of ice-free zones in Hudson Bay. Also maximum sea surface temperature >= 8ºC and salinity >=15 PSU resulted in selecting certain regions, also in Hudson Bay. The species requirements are well demonstrated and there are reliable environmental data layers for the RA area to show a match.

Background Information: Reproduction stops at salinities below 15 PSU (Josefsson and Jansson, 2011) and low temperature and ice may also be an inhibiting factor for this species as the algae requires a temperature of >8ºC for more than 4 months (Stelzer et al., 2013). It is a perennial invader without vegetative reproduction (Engelen and Santos, 2009). It is highly fecund (Norton and Deysher, 1989), producing fertile receptacles which are cast off during the summer months. These float and can survive for up to 3 months (Farnham et al., 1981). The receptacles are androgynous with self-fertilization; viable germlings are released. *Sargassum muticum* germling growth increases over the range of 5 to 25ºC and, although capable of growing at 7ºC, growth is much slower than at 17ºC. Likewise, survival was lower at 7ºC as compared to 17ºC (https://www.cabi.org/isc/datasheet/108973 and references therein).

Q6- Establishment (natural control agents) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents are unlikely to affect population growth in the RA area for *Sargassum muticum.* Although some taxa in the RA area could act as grazers, there is substantial evidence that this species of algae is generally not preferred over native algal species, due to unpalatable secondary metabolites, and grazers are not able to control growth rates. There is considerable published information on the species and its interactions and a clear mechanism for prevention of grazing.

Background Information: There is herbivory on the whole plant and/or seedling by Amphipods (e.g., *Ampithoe mea*, *Dexamine spinosa*), Gastropods (e.g., *Lacuna variegata*), and Echinoidea (e.g., *Psammechinus miliaris*) (https://www.cabi.org/isc/datasheet/108973 and references therein), although predators have a low preference for invasive *S. muticum* compared to native macroalgae. A second biotic pressure on macroalgae in temperate marine habitats is epiphyte overgrowth, which can reduce photosynthesis below the compensation point and decrease gas exchange (Strong et al., 2009).

From Engelen et al. (2015): "Grazers affected the growth rate of all macroalgae, but *Sargassum muticum* had the highest biomass growth rates with and without grazers. Similar results were obtained by Cacabelos et al. (2010), who concluded that in general native meso- and macro-grazers (*Littorina littorea*, *Aplysia punctata*, and *Paracentrotus lividus*) preferred native macroalgae to *Sargassum muticum*. From this, they suggested, perhaps prematurely, that *S. muticum* is not under substantial pressure from grazers on the Galician coast.

Brown macroalgae, including species of *Sargassum*, contain a class of polyphenolic secondary metabolites called phlorotannins (which)…deter grazing by gastropod herbivores."

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Sargassum muticum* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for widespread natural dispersal.

Background Information: *Sargassum muticum* can spread rapidly by drifting (Farnham et al., 1981). Fertile branches break off from the holdfast and float away, which is an effective system for global spread (Deysher and Norton, 1981). Detached thalli, with flotation assisted by the gas filled vesicles on the side branches, can be dispersed by currents and travel many km along coastlines (https://www.cabi.org/isc/datasheet/108973 and references therein). Adult *S. muticum* plants are also reported to be able to disperse across sediments by peripatetic "stone-walking" when the buoyancy of the plant exceeds the weight of the anchoring stone (https://www.cabi.org/isc/datasheet/108973 and references therein). In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Sargassum muticum* has the capacity for a wide range of anthropogenic dispersal in the RA area. Algae could possibly be taken up in ballast, and is known to foul ships or boats and associated structures (anchors, etc) and become entangled in fishing implements (e.g. nets, scallop dredges).

Background Information: Although primary introductions to both North America and Europe are attributed to translocation with oyster stock, the species has been known to foul vessels, which presents a potential alternative vector for long distance transfer to these new regions (https://www.cabi.org/isc/datasheet/108973). Dispersal with ships, including pleasure boats, is also a possibility through entanglement in anchor chains or propellers or hull fouling (Wallentinus, 1999).

From CABI: "*Sargassum muticum* has been known to be transported after entanglement of plants around the steering gear of vessels (Critchley et al., 1983), and as hull fouling (Abbott and Huisman, 2004). In Ireland, *S. muticum* was often found in or near harbors, mooring areas, anchorage sites and pontoons, suggesting boats as a likely vector for both arrival in Ireland from the United Kingdom or France and further distribution in Irish waters (Kraan, 2008)." (https://www.cabi.org/isc/datasheet/108973)

From Hewitt et al. (2007): "Observations and anecdotal evidence indicate that introduced species often become entangled in fishing gear such as nets and ropes, anchor ropes and chains (e.g., Carlton and Scanlon, 1985, Trowbridge, 1995, Trowbridge, 1996, 1998, Relini et al., 2000), possibly leading to further spread of these species. Macroalgal species tolerant to emersion could be successfully transported by these vectors. For example, *Caulerpa taxifolia* and *Codium fragile* ssp. *tomentosoides* survive emersion in high humidity for up to 10 and 90 days, respectively (Sant et al., 1996, Schaffelke and Deane, 2005)."

With the exception of ship/barge-related movements, the importance of other types of movements (fishing and personal vessels) in HBC is not well known.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Sargassum muticum* would have high impacts in many areas on population growth in the RA area. The species is known to impact multiple populations wherever it is invasive and strongly impacts eelgrass (*Zostera marina*), which occurs in many areas of the RA area.

Background Information: Strong competitive effects (for nutrients, light and space) by *S. muticum* on a number of native macroalgal species have been shown (White, 2010; and references therein). It decreases kelp density, seaweed cover and biomass (Sorte et al., 2010; and references therein). K-selected traits of *S. muticum* are most important to its invasive success favouring competition rather than r-selected traits that favour colonization (Engelen and Santos 2009).

From Den Hartog (1997): "Eelgrass, *Zostera marina* L., occurs mainly on sandy and muddy substrata, and the recently introduced brown alga, *Sargassum muticum* (Yendo) Fensholt is a species that adheres to solid substrata; usually they are spatially separated. However, when the bottom consists of a mixed substratum of sand, gravel and stones both species may occur together. *Sargassum muticum* hardly intrudes prosperous eelgrass beds, but in places where the beds are in decline as a consequence of the normal bed dynamics, *S. muticum* occupies almost immediately the empty spaces. There are no indications of any direct competition between *S. muticum* and *Z. marina*; however, as *S. muticum* settles in all spots where *Z. marina* has retreated, it interferes with the regeneration of the bed. In *S. muticum* stands, no germlings of *Z. marina* have been found, so in time one may expect that *S. muticum* will replace the eelgrass bed."

From CABI: "The potential effects of invasive *S. muticum* on seagrass beds has long been a concern, initially because of the observation that in British Columbia the invader occupied sheltered shallow habitats usually occupied by the seagrass *Zostera marina* (Druehl, 1973). The concern was subsequently discounted by studies that suggested that *S. muticum* required a solid substrate for attachment (North, 1973, Fletcher and Fletcher, 1975, Norton, 1977) and small stones, gravel and sand were unsuitable (Thomsen et al., 2006). Den Hartog (1997) found that *S. muticum* could replace *Z. marina* in littoral pools with unconsolidated substratum, but was unable to invade closed *Z. marina* beds on soft substrata. Investigations into the growth of *S. muticum* within *Zostera* beds in southern England found that the alga could colonize soft sediments, most likely by drifting fragments becoming trapped within the seagrass allowing settlement on the seagrass matrix in an otherwise unfavourable environment (Tweedley et al., 2008). *S. muticum* has also established in seagrass beds in Ireland (Kraan, 2008).

Studies in the subtidal zone have indicated significant effects of *S. muticum* on subtidal native assemblages through overgrowing and shading underlying species (Ambrose and Nelson, 1982, Critchley et al., 1986, Givernaud et al., 1991, Curiel et al., 1998, Stæhr et al., 2000, Britton-Simmons, 2004). *Sargassum muticum* has been reported to prevent re-establishment of the giant kelp *Macrocystis pyrifera* (Ambrose and Nelson, 1982), to displace *Halidrys siliquosa*, and to reduce cover of *Laminaria*, *Fucus* and *Codium* species through competition for hard substrate and light (Stæhr et al., 2000). In British Columbia, high native diversity was found to facilitate invader recruitment of *S. muticum* but to decrease growth and/or survivorship (White and Shurin, 2007). In Washington State, *S. muticum* had a strongly negative indirect effect on a native sea urchin by reducing the abundance of its preferred food, a native kelp species (Britton-Simmons, 2004).

In Portugal, on the highly exposed southwestern coast, *S. muticum* develops in sheltered tide pools originally inhabited by *Cystoseira humilis* (Engelen et al., 2008). However, studies have generally found little, no, or variable impact of *S. muticum* on intertidal shore assemblages (De Wreede, 1983, De Wreede, 1996, Viejo, 1997, Sanchez and Fernández, 2005, Buschbaum et al., 2006, Harries et al., 2007, Olabarria et al., 2009). In Scotland, significant differences were found in the intertidal algal and faunal communities associated with *S. muticum* and native algae (Harries et al., 2007). Reduced abundance of the dominant native alga *Dictyota dichotoma* in areas dominated by *Sargassum* was attributed to competition for space or shading, and elevated, but less diverse, faunal abundance was suggested to be a result of increased detrital input. In Northern Ireland the influence of the species varied between sites, ranging from a strong perturbation to moderate enhancement of infaunal density (Strong et al., 2006). Overall, Harries et al. (2007) concluded that, although establishment of dense areas of *S. muticum* would cause ecological change, the changes were unlikely to constitute serious ecological degradation or result in significant loss of biodiversity.

Possible positive effects of *S. muticum* on biodiversity through habitat-forming are also indicated by Buschbaum et al. (2006), who found more than 60 species of epibiont associated with the species on islands in the North Sea (German Bight). Total and average species richness were similar on rocky and sandy shores. But, whereas on rocky shores the epibiota was similar on a native fucoid, on sandy shores the only native habitat-providing species supported a different and less diverse assemblage than the invader. In the southeastern North Sea, despite its successful dispersal and increasing densities, *S. muticum* had not replaced other indigenous macroalgae, nor is there evidence of negative impacts on native species (Buschbaum et al., 2006). On the contrary, *S. muticum* may provide a suitable habitat for native species, such as epiphytic red algae, which became rare following the disappearance of its previous habitat, the European oyster beds that were lost in the 1950s due to overexploitation.

The mobile epifauna colonizing *S. muticum* in Spain was also found to be similar to that on native species (Viejo, 1999). Where *Sargassum* colonizes areas with previously low macroalgal abundance, the associated epiphytic and epifaunal communities can potentially boost secondary production through increased temporal and spatial availability of food for omnivorous fishes and decapods (Viejo, 1999). However, where *S. muticum* reduces the abundance of indigenous perennial algae, the annual loss of branches could have a negative effect on invertebrate abundance through seasonal reduction in plant biomass.

Sargassum muticum functions as a significant habitat forming species in the Wadden Sea where seagrass habitat was lost in the 1930s, and presence is correlated with increased number of native snake pipefish, *Entelurus aequoreus* (Polte and Buschbaum, 2008). Snake pipefish have been considered a red list species in the Wadden Sea (Berg et al., 1996). This association of pipefish with *S. muticum* is attributed to higher zooplankton densities with the *Sargassum* beds, which serve as prey to the pipefish (Polte and Buschbaum, 2008). The complex structure of *S. muticum* may also provide shelter for the pipefish from predation (Polte and Buschbaum, 2008). Algal drift (wrack) promotes an increase in the abundance of sandy beach macrofauna by providing a food source or shelter for small invertebrates (Rodil et al., 2008). Comparisons of macrofaunal assemblages in *S. muticum* and native algal wrack have demonstrated differences in composition and abundance, indicating that replacement of native wrack deposits by invasive wrack may have important effects on macrofaunal assemblages and ecosystem function on sandy beaches (Rodil et al., 2008)."
(https://www.cabi.org/isc/datasheet/108973)

Q10- Impact on communities **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Sargassum muticum* would have high impact in many areas on communities in the RA area. The species is known to impact multiple communities wherever it is invasive.

Background Information: There are strong competitive effects for nutrients, light and space by *S. muticum* on a number of native macroalgal species. (White, 2010; and references therein). *Sargassum muticum* can cause negative effects on native macroalgal biodiversity once it becomes abundant in its exotic range. It can shift macroalgal community structure toward lower diversity and evenness (White, 2010; and references therein), but it can have positive effects on benthic macroalgal richness by providing habitat for filamentous epiphytes and increased richness of mobile marine invertebrates such as amphipods and littorine snails by increasing available habitat through its distinct, complex structure (White, 2010; and references therein). In non-native regions, *S. muticum* attains larger sizes with lower levels of defensive polyphenolic compounds. This increase in size provides competitive superiority over native macroalgae (White, 2010).

From CABI: "The potential effects of invasive *S. muticum* on seagrass beds has long been a concern, initially because of the observation that in British Columbia the invader occupied sheltered shallow habitats usually occupied by the seagrass *Zostera marina* (Druehl, 1973). The concern was subsequently discounted by studies that suggested that *S. muticum* required a solid substrate for attachment (North, 1973, Fletcher and Fletcher, 1975, Norton, 1977) and small stones, gravel and sand were unsuitable(Thomsen et al., 2006). Den Hartog (1997) found that *S. muticum* could replace *Z. marina* in littoral pools with unconsolidated substratum, but was unable to invade closed *Z. marina* beds on soft substrata. However, on more sandy or gravelly substrata, and where beds were in decline as a consequence of normal bed dynamics, *S. muticum* could rapidly occupy available space (Den Hartog, 1997). Investigations into the growth of *S. muticum* within *Zostera* beds in southern England found that the alga could colonize soft sediments, most likely by drifting fragments becoming trapped within the seagrass allowing settlement on the seagrass matrix in an otherwise unfavourable environment (Tweedley et al., 2008). Once settled, *S. muticum* may interfere with seagrass bed regeneration. *Zostera* germlings have not been found in *S. muticum* stands (Den Hartog, 1997). *S. muticum* has also established in seagrass beds in Ireland (Kraan, 2008).

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Sargassum muticum functions as a significant habitat forming species in the Wadden Sea where seagrass habitat was lost in the 1930s, and presence is correlated with increased number of native snake pipefish, *Entelurus aequoreus* (Polte and Buschbaum, 2008). Snake pipefish have been considered a red list species in the Wadden Sea (Berg et al., 1996). This association of pipefish with *S. muticum* is attributed to higher zooplankton densities with the *Sargassum* beds, which serve as prey to the pipefish (Polte and Buschbaum, 2008). The complex structure of *S. muticum* may also provide shelter for the pipefish from predation (Polte and Buschbaum, 2008). Algal drift (wrack) promotes an increase in the abundance of sandy beach macrofauna by providing a food source or shelter for small invertebrates (Rodil et al., 2008). Comparisons of macrofaunal assemblages in *S. muticum* and native algal wrack have demonstrated differences in composition and abundance, indicating that replacement of native wrack deposits by invasive wrack may have important effects on macrofaunal assemblages and ecosystem function on sandy beaches (Rodil et al., 2008)."
(https ://www.cabi.org/isc/datasheet/108973)

Q11- Impact on habitats **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Sargassum muticum* would have high impact in many areas on habitat in the RA area. This is a structural species and it has been shown to impact various habitats (benthic, intertidal via wrack, etc.).

Background Information: In general, exotic macroalgae impact native macroalgal community structure and diversity (Nyberg, 2007) by altering the physical, chemical, and biotic characteristics of the habitat. *Sargassum muticum* has been found to have a negative effect on native species by reducing the light conditions for benthic microalgae, especially in the subtidal and lower intertidal zones (Wallentinus and Nyberg, 2007; and references therein). It can also increase nitrate concentrations (Sorte et al., 2010; and references therein). *Sargassum muticum* can act as a new substratum since it can grow up to an incredible 16 m in length in certain habitats, and can form floating mats on the sea surface (https://www.seaweed.ie/descriptions/Sargassum_muticum.php). Positive chemotaxis induced by some fucoid exudates attracts the larvae of some hydroids (e.g. *Coryne uchidai*) to inhabit *Sargassum* spp. (Nishihira, 1968, Ronowicz et al., 2013).

From Stelzer et al. (2013): “It reduces the sunlight reaching the bottom and pose a threat to the benthic community (Främmande, 2006) and it can lead to a large increase in biomass input and thus of detritus in the system (Josefsson and Jansson, 2011). The ‘bushy’ structure and high densities also have physical impacts, such as higher sedimentation rates.”

Because *S. muticum* undergoes a faster and more complete decomposition than that of the native flora it has increased the turnover and regeneration of nutrients in Limfjorden, Denmark, and thus altered the nutrient cycle (Pedersen et al., 2005). Oxygen free conditions and development of hydrogen sulphide caused by stagnation of water movement in dense stands of *S. muticum* has been observed (Karlsson et al., 1995).

From CABI: “In the southern Wadden Sea, *S. muticum* has become a significant habitat forming species in shallow subtidal areas that had become unvegetated following the decline of the seagrass *Zostera marina* during the 1930s (Polte and Buschbaum, 2008). Similar habitat modification has been noted in Northern Ireland, where extensive colonization of unvegetated soft sediments has generated a new epibenthic habitat that, in turn, has modified resident infaunal assemblages (Strong et al., 2006). Stands of *S. muticum* also caused strong temperature stratification by cooling of the water just above the sediment, and heating a thin water layer associated with the canopy. High densities of *S. muticum* can restrict water exchange with adjacent non-canopy areas, resulting in the stagnant water within the stands becoming extremely warm on sunny days (Strong et al., 2006)

The potential effects of invasive *S. muticum* on seagrass beds has long been a concern, initially because of the observation that in British Columbia the invader occupied sheltered shallow habitats usually occupied by the seagrass *Zostera marina* (Druehl, 1973). The concern was subsequently discounted by studies that suggested that *S. muticum* required a solid substrate for attachment (North, 1973, Fletcher and Fletcher, 1975, Norton, 1977) and small stones, gravel and sand were unsuitable(Thomsen et al., 2006). Den Hartog (1997) found that *S. muticum* could replace *Z. marina* in littoral pools with unconsolidated substratum, but was unable to invade closed *Z. marina* beds on soft substrata. However, on more sandy or gravelly substrata, and where beds were in decline as a consequence of normal bed dynamics, *S. muticum* could rapidly occupy available space (Den Hartog, 1997). Investigations into the growth of *S. muticum* within *Zostera* beds in southern England found that the alga could colonize soft sediments, most likely by drifting fragments becoming trapped within the seagrass allowing settlement on the seagrass matrix in an otherwise unfavourable environment (Tweedley et al., 2008). Once settled, *S. muticum* may interfere with seagrass bed regeneration. *Zostera* germlings have not been found in *S. muticum* stands (Den Hartog, 1997). *S. muticum* has also established in seagrass beds in Ireland (Kraan, 2008).

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In Portugal, on the highly exposed southwestern coast, *S. muticum* develops in sheltered tide pools originally inhabited by *Cystoseira humilis* (Engelen et al., 2008). However, studies have generally found little, no, or variable impact of *S. muticum* on intertidal shore assemblages (De Wreede, 1983, De Wreede, 1996, Viejo, 1997, Sanchez and Fernández, 2005, Buschbaum et al., 2006, Harries et al., 2007, Olabarria et al., 2009). In Scotland, significant differences were found in the intertidal algal and faunal communities associated with *S. muticum* and native algae (Harries et al., 2007). Reduced abundance of the dominant native alga *Dictyota dichotoma* in areas dominated by *Sargassum* was attributed to competition for space or shading, and elevated, but less diverse, faunal abundance was suggested to be a result of increased detrital input. In Northern Ireland the influence of the species varied between sites, ranging from a strong perturbation to moderate enhancement of infaunal density (Strong et al., 2006). Overall, Harries et al. (2007) concluded that, although establishment of dense areas of *S. muticum* would cause ecological change, the changes were unlikely to constitute serious ecological degradation or result in significant loss of biodiversity.

Possible positive effects of *S. muticum* on biodiversity through habitat-forming are also indicated by Buschbaum et al. (2006), who found more than 60 species of epibiont associated with the species on islands in the North Sea (German Bight). Total and average species richness were similar on rocky and sandy shores but, whereas on rocky shores the epibiota was similar on a native fucoid, on sandy shores the only native habitat-providing species supported a different and less diverse assemblage than the invader. In the southeastern North Sea, despite its successful dispersal and increasing densities, *S. muticum* had not replaced other indigenous macroalgae, nor is there evidence of negative impacts on native species (Buschbaum et al., 2006). On the contrary, *S. muticum* may provide a suitable habitat for native species, such as epiphytic red algae, which became rare following the disappearance of its previous habitat, the European oyster beds that were lost in the 1950s due to overexploitation.

The mobile epifauna colonizing *S. muticum* in Spain was also found to be similar to that on native species (Viejo, 1999). Where *Sargassum* colonizes areas with previously low macroalgal abundance, the associated epiphytic and epifaunal communities can potentially boost secondary production through increased temporal and spatial availability of food for omnivorous fishes and decapods (Viejo, 1999). However, where *S. muticum* reduces the abundance of indigenous perennial algae, the annual loss of branches could have a negative effect on invertebrate abundance through seasonal reduction in plant biomass.

S. muticum functions as a significant habitat forming species in the Wadden Sea where seagrass habitat was lost in the 1930s, and presence is correlated with increased number of native snake pipefish, *Entelurus aequoreus* (Polte and Buschbaum, 2008). Algal drift (wrack) promotes an increase in the abundance of sandy beach macrofauna by providing a food source or shelter for small invertebrates (Rodil et al., 2008). Comparisons of macrofaunal assemblages in *S. muticum* and native algal wrack have demonstrated differences in composition and abundance, indicating that replacement of native wrack deposits by invasive wrack may have important effects on macrofaunal assemblages and ecosystem function on sandy beaches (Rodil et al., 2008).” (https://www.cabi.org/isc/datasheet/108973)

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Sargassum muticum* would have high impacts in many areas on ecosystem function in the RA area. The species has been shown to impact functioning in various parts of the ecosystem (light levels, nutrients, intertidal and subtidal energy fluxes).

Background Information: Where *Sargassum* colonizes areas with previously low macroalgal abundance, the associated epiphytic and epifaunal communities can potentially boost secondary production through increased temporal and spatial availability of food for omnivorous fishes and decapods (Viejo, 1999). Algal drift (wrack) promotes an increase in the abundance of sandy beach macrofauna by providing a food source or shelter for small invertebrates, which may have important effects on macrofaunal assemblages and ecosystem function on sandy beaches (Rodil et al., 2008).

From Stelzer et al. (2013): “It reduces the sunlight reaching the bottom and pose a threat to the benthic community (Främmande, 2006) and it can lead to a large increase in biomass input and thus of detritus in the system (Josefsson and Jansson, 2011). The ‘bushy’ structure and high densities also have physical impacts, such as higher sedimentation rates.”

Because *S. muticum* undergoes a faster and more complete decomposition than that of the native flora it has increased the turnover and regeneration of nutrients in Limfjorden, Denmark, and thus altered the nutrient cycle (Pedersen et al., 2005). Oxygen free conditions and development of hydrogen sulphide caused by stagnation of water movement in dense stands of *S. muticum* has been observed (Karlsson et al., 1995).

From Engelen et al. (2015): “Although the dominance of *Sargassum muticum* in invaded habitats would be expected to have impacts at the ecosystem level, few studies have reported this. On sandy beaches of northern Spain, which do not represent ‘invaded habitats’ as such, detached *S. muticum* was an important food resource for the amphipod *Talitrus saltator* and to a lesser extent for the isopod *Tylos europaeus* (Rossi et al., 2010). The input of organic material from *Sargassum muticum* was not important, however, during spring (March and May), Rossi et al. (2010) attributed this to the high availability of the native macrophyte *Cystoseira baccata*. *Sargassum muticum* was one of the few species available as beach wrack during autumn and as such might sustain the population abundance of *Talitrus saltator* in a season when it normally declines (Rossi et al., 2010). Rossi et al. (2011) found that addition of *Sargassum muticum* wrack to beach sand resulted in higher 15N mobilization to sediments and macrofauna. However, as mixtures of *S. muticum* with *Fucus vesiculosus* and *Ulva* sp. were used, the contribution of *Sargassum muticum* is uncertain (Rossi et al., 2011).

Experimental manipulations in mesocosms have shown that the presence of *Sargassum muticum* can increase food web complexity (measured as connectance) as well as the proportion of species of intermediate trophic level and decrease the proportion of top predators (Salvaterra et al., 2013). Increased food web complexity was caused by the arrival of new generalist species and a redistribution of existing species to higher trophic levels (Salvaterra et al., 2013). In addition, using artificial macroalgal assemblages, Vaz-Pinto et al. (2014) demonstrated that, under controlled conditions, *S. muticum* increased respiration and production of the assemblages due to its high biomass. Overall, these studies suggest that effects of *Sargassum muticum* extend to the ecosystem level, but this area of research is at an early stage.”

From CABI: “In the southern Wadden Sea, *S. muticum* has become a significant habitat forming species in shallow subtidal areas that had become unvegetated following the decline of the seagrass *Zostera marina* during the 1930s (Polte and Buschbaum, 2008). Similar habitat modification has been noted in Northern Ireland, where extensive colonization of unvegetated soft sediments has generated a new epibenthic habitat that, in turn, has modified resident infaunal assemblages (Strong et al., 2006). Stands of *S. muticum* also caused strong temperature stratification by cooling of the water just above the sediment, and heating a thin water layer associated with the canopy. High densities of *S. muticum* can restrict water exchange with adjacent non-canopy areas, resulting in the stagnant water within the stands becoming extremely warm on sunny days (Strong et al., 2006).

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Possible positive effects of *S. muticum* on biodiversity through habitat-forming are also indicated by Buschbaum et al. (2006), who found more than 60 species of epibiont associated with the species on islands in the North Sea (German Bight). Total and average species richness were similar on rocky and sandy shores but, whereas on rocky shores the epibiota was similar on a native fucoid, on sandy shores the only native habitat-providing species supported a different and less diverse assemblage than the invader. In the southeastern North Sea, despite its successful dispersal and increasing densities, *S. muticum* had not replaced other indigenous macroalgae, nor is there evidence of negative impacts on native species (Buschbaum et al., 2006). On the contrary, *S. muticum* may provide a suitable habitat for native species, such as epiphytic red algae, which became rare following the disappearance of its previous habitat, the European oyster beds that were lost in the 1950s due to overexploitation.

The mobile epifauna colonizing *S. muticum* in Spain was also found to be similar to that on native species (Viejo, 1999). Where *Sargassum* colonizes areas with previously low macroalgal abundance, the associated epiphytic and epifaunal communities can potentially boost secondary production through increased temporal and spatial availability of food for omnivorous fishes and decapods (Viejo, 1999). However, where *S. muticum* reduces the abundance of indigenous perennial algae, the annual loss of branches could have a negative effect on invertebrate abundance through seasonal reduction in plant biomass.

Sargassum muticum functions as a significant habitat forming species in the Wadden Sea where seagrass habitat was lost in the 1930s, and presence is correlated with increased number of native snake pipefish, *Entelurus aequoreus* (Polte and Buschbaum, 2008). Snake pipefish have been considered a red list species in the Wadden Sea (Berg et al., 1996). This association of pipefish with *S. muticum* is attributed to higher zooplankton densities with the *Sargassum* beds, which serve as prey to the pipefish (Polte and Buschbaum, 2008). The complex structure of *S. muticum* may also provide shelter for the pipefish from predation (Polte and Buschbaum, 2008). Algal drift (wrack) promotes an increase in the abundance of sandy beach macrofauna by providing a food source or shelter for small invertebrates (Rodil et al., 2008). Comparisons of macrofaunal assemblages in *S. muticum* and native algal wrack have demonstrated differences in composition and abundance, indicating that replacement of native wrack deposits by invasive wrack may have important effects on macrofaunal assemblages and ecosystem function on sandy beaches (Rodil et al., 2008)."
(https://www.cabi.org/isc/datasheet/108973)

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Sargassum muticum* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. There are a wide variety of species that could be transported, but few appear to have major impacts. There is relatively limited general knowledge of effects of diseases and epiphytes.

Background Information: *Sargassum muticum* has great potential to introduce other species that are epiphytic or raft on detached thalli. A 5 metre tall plant may host an average of 3,000 animals, including foraminiferans, hydroids, flatworms, polychaete worms, leeches, snails, ostracods, cumaceans, isopods, gammarid and caprellid amphipods, opossum shrimp, euphausiid shrimp, crabs and bryozoans (Nicholson et al., 1981); while a study in southern England found 80 animal species, 52 seaweed species and 9 fungi living on or attached to *S. muticum* (Withers et al., 1975), including some small tube-building worms that may have been introduced with it. (https ://www.exoticsguide.org/sargassum_muticum)

From Egan et al. (2014): "Pathogenic pressure by bacteria might be substantial, as macroalgae have evolved a range of innate and induced defense mechanisms that have the potential to control bacterial attacks. The presence and abundance of virulence factors in marine bacteria, which have not previously been recognized as pathogens, also represents an underappreciated, opportunistic potential for disease. Given that virulence expression in opportunistic pathogens is often dependent on environmental conditions, we predict that current and future anthropogenic changes in the marine environment will lead to an increase in the occurrence of macroalgal disease."

From Kerrison et al. (2015): "Little is currently known about diseases in kelp, or seaweeds in general, although various causative agents have been implicated (Gachon et al., 2010). Problematic diseases/syndromes have been identified in large scale cultivation of kelp over the last half century (Scoggan et al., 1989) and their occurrence has increased as cultivation has intensified, with a 4-5% yield loss now reported in Korean cultivation (Gachon et al., 2010). The bacteria *Pseudoalterom* spp. And *Alteromonas* spp. Are known to be responsible for some disease (Egan et al., 2014), but in numerous cases, the agent has not been identified. The prevalence of endophytic infection is known to be high in wild kelp populations (Ellertsdottir and Peters, 1997), and so there are concerns that a) pathogens may be transplanted with seaweed stocks, infecting nearby natural seaweed beds and b) as physicochemical stress is often a trigger for outbreaks in cultivated kelp (Scoggan et al., 1989), climate change impacts such as rising seawater temperatures may in the future lead to more severe disease impacts."

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Sargassum muticum* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: No genetic effects have been observed (https://www.nobanis.org/globalassets/speciesinfo/s/sargassum-muticum/sargassum_muticum.pdf). There are no *Sargassum* species in the assessment area (Stewart and Lockhart, 2005, Mathieson et al., 2010, https://www.gbif.org, https://obis.org/).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Sargassum muticum* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some benthic species at risk that could be moderately impacted given the moderate effects of this algae on benthic ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: *Sargassum muticum* functions as a significant habitat forming species in the Wadden Sea where seagrass habitat was lost in the 1930s, and presence is correlated with increased number of native snake pipefish, *Entelurus aequoreus* (Polte and Buschbaum, 2008). Snake pipefish have been considered a red list species in the Wadden Sea (Berg et al., 1996). This association of pipefish with *S. muticum* is attributed to higher zooplankton densities with the *Sargassum* beds, which serve as prey to the pipefish (Polte and Buschbaum, 2008). The complex structure of *S. muticum* may also provide shelter for the pipefish from predation (Polte and Buschbaum, 2008).

There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through direct or indirect habitat, food web and ecosystem impacts – these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and thorny skate, *Amblyraja radiata*. Walrus populations that are at risk or of special concern may also be impacted as they feed on benthic invertebrates so may be impacted through changes in food resources in benthic habitats by this algae.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Sargassum muticum* would have high impacts in few areas or moderate impact in many areas on fished species in the RA area. There may be possible slight impacts to fisheries species. Aquaculture is not really practiced in the area.

Background Information: *Sargassum muticum* fouls propellers and fishing lines, and waterways and marinas when it becomes detached and forms large floating mats. It is also a pest on oyster beds and a nuisance to commercial fishermen, fouling their nets (https://www.exoticsguide.org/sargassum_muticum). They can overgrow in oyster reefs, directly affecting economic gains, and can literally float away with oysters (Wallentinus and Nyberg, 2007; and references therein). In James Bay, these plants offer habitat for many marine species and browse for large populations of migratory waterfowl that are hunted locally for subsistence and sport (Stewart and Lockhart, 2005). Shellfish are not cultivated in the assessment area. Natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005) and blue mussels are occasionally harvested for subsistence by Hudson Bay residents.

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Sargassum muticum* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Featured on the ISSG 100 of the World's Worst Invasive Alien Species list, it is one of eight top risk species in Europe (Nyberg and Wallentinus, 2005) (http://www.iucngisd.org/gisd/speciesname/Sargassum+muticum).

From White (2010): "Native to Japan, *S. muticum* is believed to have been introduced to Vancouver, Canada in the 1940s attached to imported Japanese oysters (*Crassostrea gigas*) and is now well established throughout the northeast Pacific (White, 2010). It expanded its range north to south-eastern Alaska (Wallentinus, 1999) and south to Baja, California (Núñez-López and Valdez, 1998). From British Columbia, Japanese oysters were transplanted to Atlantic France, unintentionally translocating *S. muticum* with them, as predicted by Druehl (1973). Introduced populations in Europe extend along the coasts of France, Scandinavia, the Baltic Sea, Helgoland, the Netherlands, the Iberian Peninsula, and into the Mediterranean from Italy and the Adriatic Sea (Thomas, 2002)."

Undaria pinnatifida (Harvey) Suringar, 1873

Phylum: Ochrophyta

Class: Phaeophyceae

Order: Laminariales

Family: Alariaceae

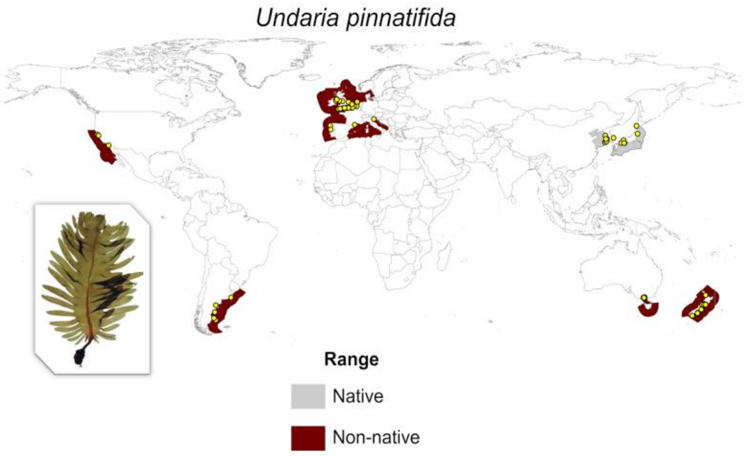

Figure 26: Ecoregions where Undaria pinnatifida is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Undaria pinnatifida occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.uiwfnr, 4 May 2017), Martin and Cuevas (2006), Primo et al. (2010), Meretta et al. (2012), Schiel and Thompson (2012), Sfriso and

Facca (2013), Pereyra et al. (2014). Picture of U. pinnatifida modified from https://www.marinebiosecurity.org.nz/undaria-pinnatifida-harvey-suringar/.

CMIST scores for *U. pinnatifida:*

Mean adjusted Likelihood of Invasion: 2.23

Mean adjusted Impact of Invasion: 2.08

Mean adjusted Overall CMIST score: 4.64

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Undaria pinnatifida* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There are no reports found from Hudson Bay (Stewart and Lockhart, 2005, Mathieson et al., 2010, Küpper et al., 2016).

Q2- Rate of introduction (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Undaria pinnatifida* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread in connected ecoregions and could be transported in ballast or through biofouling, but no studies have actually found it in vessels arriving to the Arctic. It is not established in nearby ecoregions, so natural spread is unlikely.

Background Information: *Undaria pinnatifida* is present in ports that are directly connected by shipping to Churchill and Deception Bay and could potentially be entrained and transported by these vessels in ballast water and by hull fouling (Chan et al., 2012).

Q3- Survival (suitable habitat) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Undaria pinnatifida.* There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: *Undaria* grows on rocks from the lowest inter-tidal to sub-tidal zones (https://www.cabi.org/isc/datasheet/59979). It occurs in dense, vigorous stands on benthic shores, forming thick canopy over the biota in a wide range of shores varying in exposure, from low tide level down to 15m in clear waters (http://www.iucngisd.org/gisd/speciesname/Undaria+pinnatifida). It is able to invade readily upon disturbance and then persist (Williams and Smith, 2007). *Undaria pinnatifida* also occurs abundantly in other disturbed habitats such as areas of sand scour at the base of reefs and on unstable substrata, while it occurs rarely in established macroalgal stands (Valentine and Johnson, 2003; and references therein). It grows in a wide range of wave exposures from sheltered marinas to the open coast, and extends vertically from the low intertidal to 18m depth (although it is most common between 1 and 3m depth) (http://www.iucngisd.org/gisd/speciesname/Undaria+pinnatifida). *Undaria* can grow on any hard surface including artificial substrates such as rope, pylons, buoys, the hulls of vessels, bottles, floating pontoons and plastic. On natural substrates, it inhabits stable rocky reefs, mobile cobble habitats, mudstone, and in primarily soft sediment habitats attached to hard surfaces such as shell. It can also grow on seagrass (while a small sporophyte), the shells of abalone and bivalves, invertebrates and epiphytically on other seaweeds (MFish, 2001, from GISD factsheet). The clubbed tunicate *Styela clava* may contribute to the establishment of *U. pinnatifida* (Pereyra et al., 2015). Heavy ice conditions, low salinity, and low air and water temperatures are the most important factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q4- Survival (suitable environment) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests most of the RA area offers suitable environmental conditions for *Undaria pinnatifida.* It is known to be present in cold environments and a wide region of the RA area has the adequate temperature for survival (between -1 and 12°C) combined with the salinity tolerances. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Survival ranges are <0 – >25ºC (Sporophytes), and <-1 – 29ºC (Gametophytes). Growth ranges are 3.5 – 20ºC (Sporophytes), and 10 – 24ºC (Gametophytes) (Sanderson, 1990, Minchin and Nunn, 2014). *Undaria* can tolerate a lowered salinity and grows well in some estuarine regimes, where many native marine species, especially kelps, cannot (Farrell, 2003). *Undaria* inhabits cold temperate coastal areas and grows best in waters below 12°C. Sporophytes are reported to degrade at temperatures above 20°C and die at temperatures greater than 23°C (http://www.iucngisd.org/gisd/speciesname/Undaria+pinnatifida). It tolerates a wide range of irradiance from full sunlight to very low light levels, but is unlikely to invade areas with a high freshwater input (http://www.iucngisd.org/gisd/speciesname/Undaria+pinnatifida). Nutrient supply seems to be important for the establishment and production of benthic algal communities. It is even possible that the absence of an algal community in an otherwise suitable habitat could be directly related to an inadequate supply of nutrients (Dunbar, 1968). Heavy ice conditions, low salinity, and low air and water temperatures are the most important factors to consider in attempting to determine the limitations of intertidal benthic algae (Lee, 1973).

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are sometimes available for *Undaria pinnatifida* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >=7ºC (minimum temperature for reproduction and recruitment of sporophytes) resulted in selecting some areas for a portion of the year. No areas would be expected to exceed the upper maximum temperature for reproduction. The species requirements for reproduction are well demonstrated and there are reliable environmental data layers for the RA area to show a match.

Background Information: From CABI: "Reproductive ranges : 7 – 23ºC (Sporophytes), 10 – 24ºC (Gametophytes) (Minchin and Nunn, 2014; from Sanderson, 1990). Recruitment of sporophytes takes place from 7 to 8ºC (Stuart and Brown, 1996). Sporophytes are generally intolerant of temperatures much over 20ºC and die-off at 29ºC (Peters and Breeman, 1992). *Undaria pinnatifida* has an annual heteromorphic life cycle, alternating between the diploid sporophyte, which is easily recognized and may occur for up to nine months, and a microscopic haploid gametophyte stage (Minchin and Nunn, 2014). Macroscopic sporophytes typically recruit in winter growing through spring to a length of up to 2 m. Reproduction is thought to occur during late spring–early summer, after which the plant degenerates (Valentine and Johnson, 2003). It has many characteristics of an opportunistic species, with a short lifespan, high growth rate, a high biomass invested in reproduction, small propagule size, a high number of propagules released, and a single reproductive episode (Valentine and Johnson, 2003; and references therein). It reproduces sexually and asexually. In Europe, it is thought that it might expand its range as far north as the entrance to the Barents Sea where summer temperatures may attain ~10 ºC (Ingvaldsen and Loeng, 2009), which is sufficient for reproduction. The ability of microscopic stages to go dormant at high temperatures may allow this species to persist during transport". (https://www.cabi.org/isc/datasheet/59979)

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Undaria pinnatifida.* Grazers find this species edible and there is some evidence that recruitment at the sporophyte stage may be affected by grazing. Although there is good published information on the species and its interactions, information is conflicting and some studies suggest grazers may have limited ability to control growth rates bringing into question their ability to slow population growth.

Background Information: Despite eating introduced seaweeds, native herbivores have not been documented to control invader spread (Williams and Smith, 2007; and references therein). Strikingly, even *Undaria pinnatifida* escapes herbivore control despite being highly edible (farmed for human consumption) and rapidly consumed by herbivores (Thornber et al., 2004). In the Tasman Sea, sea urchins cannot keep up with annual growth of *Undaria* and they actually facilitate its spread by consuming native perennial seaweeds and opening space for settlement (Williams and Smith, 2007; and references therein). On sea urchin barrens, grazing could not prevent canopy development of *Undaria pinnatifida* over a 30-month period, while native canopy species were not able to exceed more than 0.7% cover (Valentine and Johnson, 2003, Enge et al., 2017; and references therein). Thornber et al. (2004) suggest that grazing may limit recruitment and growth/survival of *Undaria pinnatifida*, although the effect was variable over time.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* has the capacity for a wide range of natural dispersal in the RA area. It can be dispersed by currents and drifting in different life stages (spores, sporophylls, fragments, fertile sporophytes). Spores are motile. It has several ways of natural spread that could happen in the assessment area. Good evidence for widespread natural dispersal.

Background Information: Laboratory experiments and field observations of spore dispersal confirmed that while spore dispersal is likely to be important for short-range dispersal (tens of metres), drifting sporophylls or fragments enable dispersal in the scale of hundreds of metres to kilometres (Forrest et al., 2000). Fertile sporophytes, which carry great numbers of spores, can be dispersed as drifting wrack by strong tidal currents and storm events (Minchin and Nunn, 2014). Should sporophytes attach to shells and stones, currents can drag plants as the lamina grows larger (Minchin and Nunn, 2014). Under laboratory conditions, the majority of *Undaria* spores are viable in seawater for at least 5 days, with some viable after 14 days (Forrest et al., 2000). *Undaria* spores are motile and swim at speeds of 3-8 mm/s in laboratory cultures (Sutō, 1950).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* has the capacity for a wide range of anthropogenic dispersal in the RA area. Algae could possibly be taken up in ballast, and is known to foul ships or boats and associated structures (anchors, etc) and become entangled in fishing implements (e.g. nets, scallop dredges).

Background Information: Introductions of *Undaria pinnatifida* were thought to have occurred accidentally via international shipping activity, mediated either through hull fouling or discharge of ballast water, or associated with translocation of aquaculture organisms (Perez et al., 1981). The ability of microscopic stages to go dormant at high temperatures may allow this species to persist during transport (https://www.cabi.org/isc/datasheet/59979). *Undaria pinnatifida* is present in ports that are directly connected by shipping to Churchill and Deception Bay and could potentially be entrained and transported by these vessels in ballast water and through hull fouling (Chan et al., 2012). Hull fouling and ballast transported by domestic coastal resupply and other boat traffic could spread *U. pinnatifida* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012).

From Hewitt et al. (2007): “Observations and anecdotal evidence indicate that introduced species often become entangled in fishing gear such as nets and ropes, anchor ropes and chains (e.g., Carlton and Scanlon, 1985, Trowbridge, 1995, Trowbridge, 1996, 1998, Relini et al., 2000), possibly leading to further spread of these species. Macroalgal species tolerant to emersion could be successfully transported by this vector. For example, *Caulerpa taxifolia* and *Codium fragile* ssp. *tomentosoides* survive emersion in high humidity for up to 10 and 90 days, respectively (Sant et al., 1996, Schaffelke and Deane, 2005).”

With the exception of ship-related movements, the importance of other types of movements (fishing and personal vessels) in HBC is not well known.

Q9- Impact on population (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* would have high impact in few areas or moderate impact in many areas on population growth in the RA area. There seems to be considerable information showing limited or moderate effects depending on the context.

Background Information: *Undaria pinnatifida* may decrease native seaweed richness and cover as well as competitor abundance (Sorte et al., 2010; and references therein). *Undaria* may displace multi-species macroalgal communities characterised by *Carpophyllum* according to (Battershill et al., 1998), but there are other authors that consider that there is very little evidence that *Undaria* displaced native brown seaweeds (Hay and Villouta, 1993, Hay and Sanderson, 1998). It has been demonstrated that *U. pinnatifida* produces several allelopathic substances, which inhibit settlement and germination of various other seaweeds (e.g., Asiatic red, brown and green seaweeds) (Wallentinus and Nyberg, 2007; and references therein). Its presence is associated with a dramatic decrease in species richness and diversity of native seaweeds in Patagonian Argentina (Casas et al., 2004) and it is able to competitively displace native species in sheltered to moderately exposed waters (Valentine and Johnson, 2003; and references therein). It is able to compete with native kelp species in the shallow sublittoral/infralittoral zone (Farrell and Fletcher, 2000). There is a significant negative correlation between sea urchin densities and native algae, but a significant positive correlation between sea urchins and *U. pinnatifida* (Valentine and Johnson, 2003). *Undaria*’s infestation levels and associated effects are likely to vary from place to place, and for reasons that may never be well understood. Therefore, even with compelling evidence of impacts (or lack thereof) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change (Forrest and Taylor, 2002).

From South et al. (2017): "Studies that have experimentally tested the competitive ability of *Undaria* to invade intact stands of native canopy-formers indicate that it does not ‘drive’ change, because un-manipulated canopies were generally not invaded and canopy-formers were able to recolonize areas despite the presence of *Undaria* (Valentine and Johnson, 2003, Edgar et al., 2004, Thompson and Schiel, 2012, South and Thomsen, 2016). Furthermore, invader-removal studies suggest that *Undaria* does not reduce the abundance of canopy-forming seaweeds (Valentine and Johnson, 2005, South et al., 2016, South and Thomsen, 2016). The impact of *Undaria* on recruitment of native perennial canopy forming species in Australasia appears to be weaker than that of other large invasive algae such as *Codium fragile* ssp*. tomentosoides* (Levings et al., 2002, Scheibling and Gagnon, 2006) and *Sargassum muticum* (Ambrose and Nelson, 1982, Stæhr et al., 2000, Britton-Simmons, 2004). There is, however, increasing evidence that *Undaria* can inhibit smaller seasonal or opportunistic native species (Valentine and Johnson, 2005, South et al., 2016, South and Thomsen, 2016). "

From Stuart (2003): “Colonisation by *Undaria* can result in an increase in biodiversity in areas otherwise devoid of diverse indigenous seaweed assemblages, but can also result in a decrease in biodiversity in other regions where it displaces species or results in a loss of spatial heterogeneity. Studies in Giudecca, Venice, have shown that *Undaria* competes with indigenous seaweeds for space, resulting in reduced biodiversity and reduced surface cover of *Rhodymenia ardissonei*, *Gracilaria verrucosa*, *Ulva rigida* and *Enteromorpha* spp. (Curiel et al., 1998). Colonisation by *Undaria* in New Zealand has been shown to increase biodiversity in some areas, such as the Marlborough Sounds, that are otherwise devoid of diverse indigenous seaweed. The increase in biodiversity is attributed to an increase in habitat complexity (Battershill et al., 1998). This same study found that subcanopy assemblages under stands of *Carpophyllum* spp. comprised a patchy mosaic of turfing and bushy seaweeds (e.g. *Ulva*, *Dictyota*, *Gigartina* and *Gelidium* spp.) together with encrusting and articulate coralline algae and bryozoans. As the density of *Undaria* increased, however, this community appeared to be displaced by solitary sea-squirts, tube worms and hydroids. While this may appear to indicate significant impacts, this study does not have the benefit of observations made before the seaweed beds were invaded by *Undaria*. Hence, there is an assumption that *Carpophyllum* spp. once inhabited patches where *Undaria* was present. This may not have been the case, and it is equally possible that the observed distribution of species reflected pre-existing factors unrelated to the presence or absence of *Undaria*. Similar studies conducted in the Mercury Passage, Tasmania found a reduction in the diversity and abundance of fauna associated with a canopy of *Undaria* compared with an adjacent canopy of indigenous species (Tasmanian Aquaculture and Fisheries Institute, 2000).”

Q10- Impact on communities (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* would have high impact in few areas or moderate impact in many areas on communities in the RA area. There seems to be considerable information showing limited or moderate effects depending on the context.

Background Information: Its presence is associated with a dramatic decrease in species richness and diversity of native seaweeds in Patagonian Argentina (Casas et al., 2004). *Undaria*’s infestation levels and associated effects are likely to vary from place to place, for reasons that may never be well understood. Hence even with compelling evidence of impacts (or lack thereof) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change (Forrest and Taylor, 2002).

From South et al. (2017): "Reported impacts of *Undaria* on community metrics such as species richness, evenness, or diversity have generally been neutral, regardless of the trophic level studied. To date, there is little evidence to support that *Undaria* displaces native canopy-forming species (Forrest and Taylor, 2002, Valentine and Johnson, 2005, South et al., 2016, South and Thomsen, 2016). Rather, it seems likely that *Undaria* has a facilitative effect on invertebrate abundances, through the provision of additional resources, that could have cascading implications for local food-web structures (Schmidt and Scheibling, 2006). Some native taxa such as abalone (*Haliotis iris*), amphipods (*Aora typica*) and turbonid snails (*Cookia sulcata* and *Lunella smaragda*) consume *Undaria* readily (Thompson, 2004, Jiménez et al., 2015), although the common epifaunal isopod *Badedotea elongata* rejected it as a food source in feeding trials (Jiménez et al., 2015). On the other hand, little is known about the epifaunal communities associated with algal species that are displaced by *Undaria* (e.g. filamentous algae and *Lophothamnion hirtum*, (Valentine and Johnson, 2005), (South et al., 2016; respectively). Therefore, the impact of *Undaria* on food-webs is likely to be complex, with many areas of uncertainties that warrant further investigation.

The export of kelp detritus can be an important trophic subsidy across ecosystems (Wernberg et al., 2006, Vanderklift and Wernberg, 2008, Krumhansl and Scheibling, 2012b). Invasive macroalgae can modify such subsidies and, consequently, local assemblages of detritivores (Krumhansl and Scheibling, 2012a) and possibly filter-feeders. It is possible that much of the negatively buoyant *Undaria* biomass detached from heavily-infested shores in southern New Zealand is exported to deeper waters offshore. Indeed, sunken *Undaria* biomass can reduce subtidal reef complexity by filling cracks and crevices sufficiently enough to reduce habitat for reef fish in Argentina (Irigoyen et al., 2011). However, the detritivore communities associated with subtidal detritus from *Undaria* have not been studied in Australasia or elsewhere. Where *Undaria* does get washed ashore it can provide a considerable trophic subsidy (Suárez-Jiménez et al., 2017). For example, the talitrid amphipod *Bellorchestia quoyana* consumes *Undaria* at similar rates to other algae that are commonly stranded on sandy beaches (Suárez-Jiménez et al., 2017). It therefore seems likely that *Undaria* represents a significant addition to coastal food-webs. However, more research is warranted to determine the extent and scale of trophic provisioning associated with this invader. Overall, the impacts of *Undaria* are not well understood due to a deficit of studies, but it seems likely that its advent has modified patterns of biogenic habitat provision and trophic subsidies locally and across ecosystems, despite its seeming inability to displace many native taxa. However, it should be noted that research has focused on a relatively small number of response variables, whereas impacts might be restricted to less easily quantified responses. These include reproductive capacity (Lyons and Scheibling, 2007), growth (Carnell and Keough, 2014) and sublethal stress (Terlizzi et al., 2011), all of which could compromise the resilience of coastal ecological communities and therefore warrant further research."

From Stuart (2003): “Colonisation by *Undaria* can result in an increase in biodiversity in areas otherwise devoid of diverse indigenous seaweed assemblages, but can also result in a decrease in biodiversity in other regions where it displaces species or results in a loss of spatial heterogeneity. Studies in Giudecca, Venice, have shown that *Undaria* competes with indigenous seaweeds for space, resulting in reduced biodiversity and reduced surface cover of *Rhodymenia ardissonei*, *Gracilaria verrucosa*, *Ulva rigida* and *Enteromorpha* spp. (Curiel et al., 1998). Colonisation by *Undaria* in New Zealand has been shown to increase biodiversity in some areas, such as the Marlborough Sounds, that are otherwise devoid of diverse indigenous seaweed. The increase in biodiversity is attributed to an increase in habitat complexity (Battershill et al., 1998). This same study found that subcanopy assemblages under stands of *Carpophyllum* spp. comprised a patchy mosaic of turfing and bushy seaweeds (e.g. *Ulva*, *Dictyota*, *Gigartina* and *Gelidium* spp.) together with encrusting and articulate coralline algae and bryozoans. As the density of *Undaria* increased, however, this community appeared to be displaced by solitary sea-squirts, tube worms and hydroids. While this may appear to indicate significant impacts, this study does not have the benefit of observations made before the seaweed beds were invaded by *Undaria*. Hence, there is an assumption that *Carpophyllum* spp. once inhabited patches where *Undaria* was present. This may not have been the case, and it is equally possible that the observed distribution of species reflected pre-existing factors unrelated to the presence or absence of *Undaria*. Similar studies conducted in the Mercury Passage, Tasmania found a reduction in the diversity and abundance of fauna associated with a canopy of *Undaria* compared with an adjacent canopy of indigenous species (Tasmanian Aquaculture and Fisheries Institute, 2000).”

Q11- Impact on habitats **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Undaria pinnatifida* would have high impact in few areas or moderate impact in many areas on habitat in the RA area.

Background Information: This type of algae is effective in providing habitats, nursery areas and protective cover for many species (Oakley, 2007). It can be an ecological engineer, i.e. creating a habitat that is suitable for other invasive species to settle (https://www.cabi.org/isc/datasheet/59979). *Undaria*'s infestation levels and associated effects are likely to vary from place to place, and for reasons that may never be well understood. Hence even with compelling evidence of impacts (or lack thereof) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change (Forrest and Taylor, 2002).

From South et al. (2017): "Reported impacts of *Undaria* on community metrics such as species richness, evenness, or diversity have generally been neutral, regardless of the trophic level studied. To date, there is little evidence to support that *Undaria* displaces native canopy-forming species (Forrest and Taylor, 2002, Valentine and Johnson, 2005, South et al., 2016, South and Thomsen, 2016).

The export of kelp detritus can be an important trophic subsidy across ecosystems (Wernberg et al., 2006, Vanderklift and Wernberg, 2008, Krumhansl and Scheibling, 2012b). Invasive macroalgae can modify such subsidies and, consequently, local assemblages of detritivores (Krumhansl and Scheibling, 2012a) and possibly filter-feeders. It is possible that much of the negatively buoyant *Undaria* biomass detached from heavily-infested shores in southern New Zealand is exported to deeper waters offshore. Indeed, sunken *Undaria* biomass can reduce subtidal reef complexity by filling cracks and crevices sufficiently enough to reduce habitat for reef fish in Argentina (Irigoyen et al., 2011). However, the detritivore communities associated with subtidal detritus from *Undaria* have not been studied in Australasia or elsewhere. Where *Undaria* does get washed ashore it can provide a considerable trophic subsidy (Suárez-Jiménez et al., 2017). Overall, the impacts of *Undaria* are not well understood due to a deficit of studies, but it seems likely that its advent has modified patterns of biogenic habitat provision and trophic subsidies locally and across ecosystems, despite its seeming inability to displace many native taxa. However, it should be noted that research has focused on a relatively small number of response variables, whereas impacts might be restricted to less easily quantified responses. These include reproductive capacity (Lyons and Scheibling, 2007), growth (Carnell and Keough, 2014) and sublethal stress (Terlizzi et al., 2011), all of which could compromise the resilience of coastal ecological communities and therefore warrant further research."

Q12- Impact on ecosystem function (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area.

Background Information: *Undaria pinnatifida* poses a threat to natural ecosystems and associated fisheries through displacement of native species via the development of 'mono-specific' *Undaria* stands (Forrest and Taylor, 2002; and references therein). *Undaria*'s infestation levels and associated effects are likely to vary from place to place, for reasons that may never be well understood. Hence even with compelling evidence of impacts (or lack thereof) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change (Forrest and Taylor, 2002).

From South et al. (2017): "Reported impacts of *Undaria* on community metrics such as species richness, evenness, or diversity have generally been neutral, regardless of the trophic level studied. To date, there is little evidence to support that *Undaria* displaces native canopy-forming species (Forrest and Taylor, 2002, Valentine and Johnson, 2005, South et al., 2016, South and Thomsen, 2016). Rather, it seems likely that *Undaria* has a facilitative effect on invertebrate abundances, through the provision of additional resources, that could have cascading implications for local food-web structures (Schmidt and Scheibling, 2006). Some native taxa such as abalone (*Haliotis iris*), amphipods (*Aora typica*) and turbonid snails (*Cookia sulcata* and *Lunella smaragda*) consume *Undaria* readily (Thompson, 2004, Jiménez et al., 2015), although the common epifaunal isopod *Badedotea elongata* rejected it as a food source in feeding trials (Jiménez et al., 2015). On the other hand, little is known about the epifaunal communities associated with algal species that are displaced by *Undaria* (e.g. filamentous algae and *Lophothamnion hirtum*, (Valentine and Johnson, 2005), (South et al., 2016; respectively). Therefore, the impact of *Undaria* on food-webs is likely to be complex, with many areas of uncertainties that warrant further investigation.

The export of kelp detritus can be an important trophic subsidy across ecosystems (Wernberg et al., 2006, Vanderklift and Wernberg, 2008, Krumhansl and Scheibling, 2012b). Invasive macroalgae can modify such subsidies and, consequently, local assemblages of detritivores (Krumhansl and Scheibling, 2012a) and possibly filter-feeders. It is possible that much of the negatively buoyant *Undaria* biomass detached from heavily-infested shores in southern New Zealand is exported to deeper waters offshore. Indeed, sunken *Undaria* biomass can reduce subtidal reef complexity by filling cracks and crevices sufficiently enough to reduce habitat for reef fish in Argentina (Irigoyen et al., 2011). However, the detritivore communities associated with subtidal detritus from *Undaria* have not been studied in Australasia or elsewhere. Where *Undaria* does get washed ashore it can provide a considerable trophic subsidy (Suárez-Jiménez et al., 2017). For example, the talitrid amphipod *Bellorchestia quoyana* consumes *Undaria* at similar rates to other algae that are commonly stranded on sandy beaches (Suárez-Jiménez et al., 2017). It therefore seems likely that *Undaria* represents a significant addition to coastal food-webs. However, more research is warranted to determine the extent and scale of trophic provisioning associated with this invader. Overall, the impacts of *Undaria* are not well understood due to a deficit of studies, but it seems likely that its advent has modified patterns of biogenic habitat provision and trophic subsidies locally and across ecosystems, despite its seeming inability to displace many native taxa. However, it should be noted that research has focused on a relatively small number of response variables, whereas impacts might be restricted to less easily quantified responses. These include reproductive capacity (Lyons and Scheibling, 2007), growth (Carnell and Keough, 2014) and sublethal stress (Terlizzi et al., 2011), all of which could compromise the resilience of coastal ecological communities and therefore warrant further research."

Q13- Associated diseases, parasites, and travellers (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* would have high impact in many areas associated with diseases, parasites or travelers in the RA area. A wide variety of species, including the invasive *Membranipora membranacea*, could be transported based on general patterns in other seaweeds and specific information on endophytes of *Undaria* (*L. aecidioides*). These endophytes are well documented to infect several kelps in the same genus as native species known in the RA area including *Agarum*, *Alaria*, *Laminaria* and cause reduced strength and breakage of the thalli.

Background Information: From Skriptsova and Kalita (2020): "The brown endophytic alga *Laminariocolax aecidioides* (Ectocarpales: Chordariaceae) growing in *Undaria pinnatifida* was found in Ussuriysky Bay (Peter the Great Bay, Sea of Japan) for the first time. This finding is the first record of the species in the Russian Far-Eastern seas. *Laminariocolax aecidioides* is known as a highly pathogenic species that causes morphological anomalies of the host alga. This often leads to a decrease in the strength of the thalli and to tearing off of the blades during storms, as was observed in January 1990 off the southwestern coast of Norway (Lein et al., 1991). Kelp infection with the endophyte *L. aecidioides* is a serious problem in kelp cultivation, as this infection reduces the commercial quality of the products (Yoshida, 1979). This alga is distributed mainly in the North Atlantic (including the Baltic Sea), along the northwestern coast of North America, off the coast of Argentina, and in the Arctic (the White Sea, Spitsbergen Island) (Guiry and Guiry, 2018). In Asian Pacific waters, *L. aecidioides* occurs only off the coasts of Japan (Yoshida, 1979, Yoshida et al., 2015). Typical hosts of this species are laminarian algae of the genera *Agarum*, *Undaria*, *Alaria*, *Laminaria*, and *Hedophyllum* (Edelstein and McLachlan, 1968, Yoshida, 1979, Abbott et al., 1992, Peters and Schaffelke, 1996, Burkhardt and Peters, 1998, Gauna et al., 2009). The species was probably brought into Peter the Great Bay from Japan with ship ballast waters."

From Egan et al. (2014): "Pathogenic pressure by bacteria might be substantial, as macroalgae have evolved a range of innate and induced defense mechanisms that have the potential to control bacterial attacks. The presence and abundance of virulence factors in marine bacteria, which have not previously been recognized as pathogens, also represents an underappreciated, opportunistic potential for disease. Given that virulence expression in opportunistic pathogens is often dependent on environmental conditions, we predict that current and future anthropogenic changes in the marine environment will lead to an increase in the occurrence of macroalgal disease."

From Kerrison et al. (2015): "Little is currently known about diseases in kelp, or seaweeds in general, although various causative agents have been implicated (Gachon et al., 2010). Problematic diseases/syndromes have been identified in large scale cultivation of kelp over the last half century (Scoggan et al., 1989) and their occurrence has increased as cultivation has intensified, with a 4-5% yield loss now reported in Korean cultivation (Gachon et al., 2010). The bacteria *Pseudoalterom* spp. and *Alteromonas* spp. are known to be responsible for some disease (Egan et al., 2014), but in numerous cases, the agent has not been identified. The prevalence of endophytic infection is known to be high in wild kelp populations (Ellertsdottir and Peters, 1997), and so there are concerns that a) pathogens may be transplanted with seaweed stocks, infecting nearby natural seaweed beds and b) as physicochemical stress is often a trigger for outbreaks in cultivated kelp (Scoggan et al., 1989), climate change impacts such as rising seawater temperatures may in the future lead to more severe disease impacts."

Undaria pinnatifida can also be a host for invasive *Membranipora membranacea* (Arnold et al., 2016), which has well-known impacts on native kelp bed ecosystems.

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: There is no evidence of other *Undaria* species present in the assessment area (Mathieson et al., 2010) (https://www.gbif.org, https://obis.org/)

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Undaria pinnatifida* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some benthic species at risk that could be moderately impacted given the moderate effects of this algae on benthic ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. There are several at risk or special concern species of fish that could be affected through habitat, food web and ecosystem impacts - these include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; thorny skate, *Amblyraja radiata*. Walrus populations at risk or of special concern may also be impacted as they feed on benthic invertebrates so may be impacted through changes in food resources in benthic habitats by this algae.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Undaria pinnatifida* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. There may be possible slight impacts to fisheries species. Aquaculture is not really practiced in the area.

Background Information: *Undaria pinnatifida* can overgrow mussel beds and oyster reefs, directly affecting economic gains in some areas (Wallentinus and Nyberg, 2007; and references therein). Shellfish are not cultivated in the assessment area, but natural populations of blue mussel (*Mytilus edulis*) and Iceland scallop (*Chlamys islandica*) have been harvested by commercial test fisheries in the Belcher Islands and along the Nunavik coast, respectively (Stewart and Lockhart, 2005). Blue mussels are occasionally harvested for subsistence by Hudson Bay residents. It may pose a threat to natural ecosystems and associated fisheries through displacement of native species via the development of ‘mono-specific’ *Undaria* stands (Forrest and Taylor, 2002; and references therein). In James Bay, eelgrass offers habitat for many marine species and browse for large populations of migratory waterfowl that are hunted locally for subsistence and sport (Stewart and Lockhart, 2005). *Undaria*’s infestation levels and associated effects are likely to vary from place to place, for reasons that may never be well understood. Hence even with compelling evidence of impacts (or lack thereof) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change (Forrest and Taylor, 2002). It has the potential to become a problem for marine farms by increasing labor and harvesting costs due to fouling problems (NIMPIS 2002, from GISD factsheet).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Undaria pinnatifida* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: *Undaria pinnatifida* is one of five top risk species in Europe (Nyberg and Wallentinus, 2005) (http://www.iucngisd.org/gisd/speciesname/Undaria+pinnatifida). It has been traditionally cultivated in its native range of eastern Asia, including Japan, Korea and China. However, in recent decades it has arisen as an invasive threat in Europe, North America and New Zealand among other places, due to human-mediated transport (Murphy et al., 2016; and references therein). While it was intentionally introduced to the Atlantic Coast of Europe in 1983 (Floc'h et al., 1991), introductions to other areas are all thought to have occurred accidentally via international shipping activity, mediated either through hull fouling or discharge of ballast water, or associated with translocation of aquaculture organisms (Perez et al., 1981). It was found attached to Japanese dock debris washed ashore in Oregon, USA after Tohoku tsunami in 2011 (https://www.cabi.org/isc/datasheet/59979).

The kelp *Undaria pinnatifida* is one of the most successful marine invasive species worldwide, and is widely regarded as one of the worst (South et al., 2017).

ZOOPLANKTON – COPEPODS

Acartia (Acanthacartia) tonsa Dana, 1849

Phylum: Arthropoda

Class: Hexanauplia

Order: Calanoida

Family: Acartidae

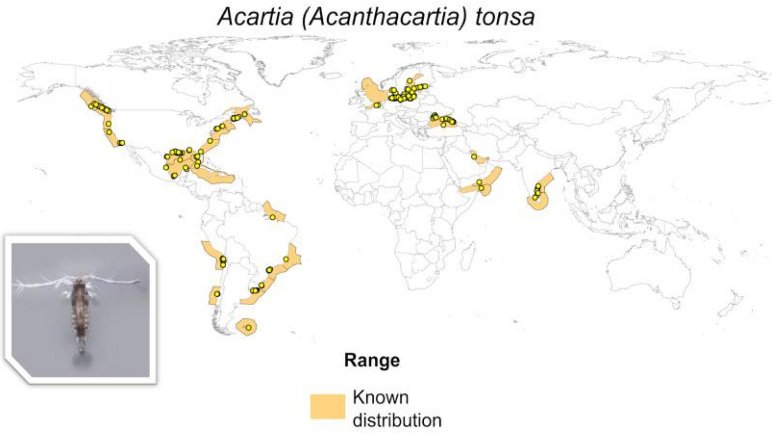

Figure 27: Ecoregions where Acartia (Acanthacartia) tonsa is distributed (orange). As it is a cryptic species, its origin and species status remain uncertain. These regions are only representative of their main known distribution, and occurrence points (in yellow) have not been collected in an exhaustive manner. Acartia (Acanthacartia) tonsa occurrence points were obtained from OBIS (https://obis.org/) and GBIF.org (http://doi.org/10.15468/dl.sumqig, 3 May 2017). Picture of A. tonsa modified from http://sio-legacy.ucsd.edu/zooplanktonguide/species/acartia-acanthacartia-tonsa.

CMIST scores for *A. tonsa:*

Mean adjusted Likelihood of Invasion: 2.48

Mean adjusted Impact of Invasion: 2.49

Mean adjusted Overall CMIST score: 6.16

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Acartia tonsa* is not established in the RA area. It was found in eDNA samples, but information is not reliable. Survey effort is low relative to the size of the RA area.

Background Information: There are no reports found from Hudson Bay (Stewart and Lockhart, 2005, Brown et al., 2016, Chain et al., 2016, Dispas, 2019). Genetic material for this species was found in eDNA samples in Churchill by Lacoursière-Roussel et al. (2018), however this was most likely misidentified according to the following explanation from the paper: "The only potential invaders detected, the Arthropoda *Acartia tonsa*, was found with the COI1 primers in Churchill (64 reads averaging 99.4% identity with the sequence references). This species was previously recorded in ballast water in ports connected to Churchill and is considered a potential invader (Chan et al., 2012). However, COI sequences in BOLD assigned to *A. tonsa* are not monophyletic and several are indistinguishable from sequences assigned to the native *A. hudsonica*, suggesting misidentification of some *Acartia* specimens in BOLD."

Q2- Rate of introduction (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Acartia tonsa* could arrive frequently in high numbers in the RA area. The species has been found on arriving vessels and is present in a number of connected port ecoregions.

Background Information: Ships visiting Churchill from other areas of Hudson Bay and Canada, and from the UK can carry live *Acartia* spp. In their ballast water and on fouled hulls (Chan et al., 2015). Dispas (2019) reported multiple specimens of *A. tonsa* in the ballast water of ships visiting Churchill and Deception Bay (see also Chan et al., 2012).

Q3- Survival (suitable habitat) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable habitat for A*cartia tonsa.* There is very wide habitat available (estuarine, marine and coastal) and no specific habitat requirement that can decrease the extension of habitat in the assessment area. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support this classification.

Background Information: This is a coastal and estuarine species (https://www.nobanis.org/globalassets/speciesinfo/a/acartia-tonsa/acartia-tonsa.pdf).

From Andersen Borg (2009): “This opportunistic species is more or less confined to coastal waters with high food concentrations and relatively high temperatures (Conover, 1956, Paffenhöfer and Stearns, 1988, Lawrence et al., 2004). They have a wide distribution in temperate and subtropical estuarine, and neritic waters of the western Atlantic Ocean and the Indo-Pacific (Mauchline, 1998, Razouls et al., 2005-2009).”

This is a marine, brackish and estuarine species (Molnar et al., 2008; and references therein).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Acartia tonsa.* It can withstand temperatures of -1°C and it has a wide salinity tolerance (in the lab and field). Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: *Acartia tonsa* has impressive physiological tolerances which enables it to sustain populations in estuaries under conditions that most other pelagic copepod species cannot tolerate (Andersen Borg, 2009). It is a euryhaline and eurythermal species with a very large range of tolerance measured by both in situ and laboratory experiments (Cervetto et al., 1999). The species does best at salinities between 15 and 22 PSU, but in laboratory experiments has survived everything from 0 to 77 PSU (Selander, 2005).

From Chaalali et al. (2013): “Temperature is the factor that controls the geographical distribution. The optimal value of temperature is around 17–25°C with modeling approaches, and an abundance reduction being estimated under a temperature value of 10-13°C (Chaalali et al., 2013). In the lab, it is able to tolerate a wide range of temperatures between -1°C and 32°C (Gonzalez, 1974). The species is also found in places such as Rhode Island where temperatures were about -1°C (Gonzalez, 1974). Under lab conditions, impressive tolerances in a range between 1 and 72 PSU with an optimal adaptation of the species between 15 and 22 PSU for copepodites (Cervetto et al., 1999). This is in accordance with observations (Tester and Turner, 1990).”

Ware et al. (2016) modelled habitat suitability for this species and predicted that although the species is not predicted to have suitable habitat in the RA under current conditions, a moderate amount of suitable habitat would be available under future conditions.

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Acartia tonsa* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >= 10°C (minimum required for reproduction) together with <= 25 PSU (required for reproduction and nauplii stages) resulted in selecting certain areas of Hudson Bay. This is also reinforced by the fact that it can produce resting eggs until conditions are favorable. The species requirements are well demonstrated and there are reliable environmental data layers for the RA area to show a match. Previous modelling by Ware et al. (2016) suggests a lack of suitable habitat under current conditions.

Background Information: A water temperature of at least +10 °C is required for successful reproduction (Selander, 2005).

From Andersen Borg (2009): "*Acartia tonsa* has the ability to produce resting eggs: diapause eggs (Castro-Longoria, 2001), eggs may enter into quiescence as a response to a hostile environment (Holmstrup et al., 2006)."

From Chaalali et al. (2013): "It may produce diapause eggs at a thermal threshold experimentally estimated at about 10–13°C (Zillioux and Gonzalez, 1972). *Acartia tonsa* has a maximal salinity threshold of 25 PSU for nauplii stages. This upper value would limit the colonization of marine waters by the early stages of development of *A. tonsa*, these stages being more sensitive than the adults to changes in salinity (Tester and Turner, 1990). Increases in eggs hatching success were observed for salinities between 0 and 17 PSU, a maximum being observed between 17 and 25 PSU (Peck and Holste, 2006). Egg hatching in *A. tonsa* was only reduced at the lowest salinity (2 PSU) (Andersen Borg, 2009). Extreme changes in salinity were needed to cause significant mortality of *A. tonsa* in the field, but its feeding activity could be severely reduced by salinity changes likely to occur in estuaries (Andersen Borg, 2009)."

Ware et al. (2016) modelled habitat suitability for this species and predicted that although the species is not predicted to have suitable habitat in the RA under current conditions, a moderate amount of suitable habitat would be available under future conditions.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Acartia tonsa.* There are closely related species in the RA that could act as predators and at least one fish predator of the genus *Clupea* that occurs within the RA area (www.gbif.org). There is limited information on predators and no information on whether or not parasites/diseases would be present in the RA area or transported with the host species in vectors.

Background Information: Stomach contents of field-collected scyphomedusa of *Chrysaora quinquecirrha* fed selectively on the calanoid copepod *Acartia tonsa* along the east coast in the northern US (Suchman and Sullivan, 1998). *Acartia* spp. Are also known to be the prey of several fish (e.g., sprat and herring, the latter of which occurs in the assessment region) (Möllmann et al., 2004, Möllmann et al., 2005).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Acartia tonsa* has the capacity for a wide range of natural dispersal in the RA area.

Background Information: *Acartia tonsa* can produce highly resistant diapause eggs as well as eggs that can be induced into quiescence in response to unfavorable conditions (Holmstrup et al., 2006). Since it can develop resistant eggs, these can be dispersed naturally to more areas and there is good evidence for widespread potential.

From Drillet et al. (2011): The production of delayed hatching eggs (DHE) reduces the risk of egg hatching at unfavorable (low-food) conditions and allows the females to spread their offspring without taking the risk associated with deep burial of eggs in the sediment. Production of DHE also increases the spatial spreading potential, particularly in areas where ocean currents are significant.

In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Acartia tonsa* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is much solid evidence for wide range dispersal through uptake in ballast and biofouling on vessels moving within the RA area.

Background Information: Accidental introductions with ballast water are known for the Black Sea, Mediterranean and Baltic Sea (Gubanova, 2000). The species has been found in ballast water samples in ships going to the Svalbard region (Ware et al., 2016). Chan et al. (2012) highlight this species as being present in ports that are connected to Canadian Arctic ports (Churchill and Deception Bay) through hull fouling and ballast water. Ballast water transported by domestic coastal resupply and other boat traffic, could spread *A. tonsa* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Ships visiting Churchill from other areas of Hudson Bay and Canada, and from the UK were found carrying live *Acartia* spp. In their ballast water and on fouled hulls (Chan et al., 2015). Dispas (2019) also reported multiple specimens of *A. tonsa* in the ballast water of ships visiting Churchill and Deception Bay. Chan et al. (2012) suggests that there is considerable domestic “Arctic direct” ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread. The production of DHE gives *A. tonsa* a great potential to invade new habitats via egg spreading and ballast water.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Acartia tonsa* would have high impacts in many areas on population growth in the RA area. There is good evidence that this species eats a wide variety of prey, exhibits multiple feeding modes, and can impact their population growth. There are demonstrated impacts, including on two species known to occur in the RA area.

Background Information: *Acartia tonsa* is omnivorous and capable of feeding on both phytoplankton and protozoan prey (Jonsson and Tiselius, 1990). It caused a phenological shift in the native copepod *Acartia bifilosa* production period, being advanced one month earlier in the year in SW Europe (David et al., 2007). It may dominate zooplankton communities due to competition (Katsanevakis et al., 2014) and is likely to act as a food competitor with co-occurring congeners in Europe (Andersen Borg, 2009). High abundances can affect the abundances of autochthonous congeneric species such as *Acartia bifilosa* (David et al., 2007).

They are an important food source for many commercial fish species (Sullivan et al., 2007). Several studies indicate they aggregate near the ocean floor during the day and rise closer to the surface at night. This behavior likely helps *A. tonsa* avoid predators who rely on vision to locate and capture prey (Fulton, 1984).

From NOBANIS: “*Acartia tonsa* feeds on nauplii of other copepods (Tackx and Polk, 1982) and on phytoplankton, including diatoms as well as flagellates. Cyanobacteria, however, are not eaten in measurable amounts (Schmidt and Jónasdóttir, 1997). It can switch between two different feeding methods, suspension feeding on non-motile prey, and actively capturing motile prey. When phytoplankton concentration is high, suspension feeding dominates (Kiørboe et al., 1996), but they strongly prefer motile prey such as ciliates and flagellates to non-motile prey such as diatoms (Sommer, 2009). The lower size limit for captured and ingested particles is about 2-4 µm (Berggreen et al., 1988).

Competition with other copepods, especially congenerics appears to be the only documented impact of *A. tonsa*. In the Black Sea it replaced the native, stenothermic *Acartia latisetosa* (Kritchagin, 1873) during the 1970s (Gubanova, 2000). In southwestern France it is seasonally replacing *Acartia bifilosa* in low salinity waters (David et al., 2007). This was attributed to intrusion of saline water plus increased water temperature in connection with a nuclear power plant, and furthermore, the change in zooplankton composition was suspected to impact production of shrimp and fish production. In a Spanish estuary with relatively high salinity, *A. tonsa* had a negative impact on *Acartia clausi* (Aravena et al., 2009). Grazing impact of *A. tonsa* appears minor; in a Mediterranean lagoon it consumed less than 1% of the phytoplankton production (Cervetto et al., 1995). A positive impact is that *A. tonsa* has been used to produce live feed for aquaculture organisms (Marchus and Wilcox, 2007, Sørensen et al., 2007).” (https://www.nobanis.org/globalassets/speciesinfo/a/acartia-tonsa/acartia-tonsa.pdf)

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Acartia tonsa* would have high impacts in many areas on communities in the RA area. There is good evidence that this species eats a wide variety of prey and can impact their population growth, and has demonstrated impacts including on two species known to occur in the RA area.

Background Information: *Acartia tonsa* is omnivorous and capable of feeding on both phytoplankton and protozoan prey (Jonsson and Tiselius, 1990). It caused a phenological shift in the native copepod *Acartia bifilosa* production period, advancing this one month earlier in the year in SW Europe (David et al., 2007). The species dominates zooplankton communities due to competition (Katsanevakis et al., 2014). It is likely to act as a food competitor with co-occurring congeners in Europe (Andersen Borg, 2009). High abundances can affect the abundances of autochthonous congeneric species such as *Acartia bifilosa* (David et al., 2007).

From Wikipedia: “They are an important food source for many commercial fish species (Sullivan et al., 2007). Several studies indicate they aggregate near the ocean floor during the day and rise closer to the surface at night. This behavior likely helps *A. tonsa* avoid predators who rely on vision to locate and capture prey (Fulton, 1984).” (https://en.wikipedia.org/wiki/Acartia_tonsa)

From NOBANIS: “*Acartia tonsa* feeds on nauplii of other copepods (Tackx and Polk, 1982) and on phytoplankton, including diatoms as well as flagellates. Cyanobacteria, however, are not eaten in measurable amounts (Schmidt and Jónasdóttir, 1997). It can switch between two different feeding methods, suspension feeding on non-motile prey, and actively capturing motile prey. When phytoplankton concentration is high, suspension feeding dominates (Kiørboe et al., 1996), but they strongly prefer motile prey such as ciliates and flagellates to non-motile prey such as diatoms (Sommer, 2009). The lower size limit for captured and ingested particles is about 2-4 µm (Berggreen et al., 1988).

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tonsa had a negative impact on *Acartia clausi* (Aravena et al., 2009). Grazing impact of *A. tonsa* appears minor; in a Mediterranean lagoon it consumed less than 1% of the phytoplankton production (Cervetto et al., 1995). A positive impact is that *A. tonsa* has been used to produce live feed for aquaculture organisms (Marchus and Wilcox, 2007, Sørensen et al., 2007)." (https://www.nobanis.org/globalassets/speciesinfo/a/acartia-tonsa/acartia-tonsa.pdf)

Q11- Impact on habitats (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest A*cartia tonsa* would have high impact in few areas or moderate impact in many areas on habitat in the RA area. This species is not known to have a physical impact on the water column per se – only on the things within it. There is no evidence of increasing water clarity, etc., but the possibility exists.

Background Information: Due to its high abundances and grazing abilities, *A. tonsa* can change energy and matter flows between pelagic and benthic systems (Leppäkoski et al., 2002). Although this is more of an ecosystem impact and there are no studies suggesting direct habitat impacts, there is a possibility of impacts on water clarity as seen in other grazers.

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests A*cartia tonsa* would have high impacts in many areas on ecosystem function in the RA area. There is abundant information on potential impacts to various ecosystem components.

Background Information: *Acartia* spp. (including *A. tonsa*) constitute significant prey for fish in the Baltic (Möllmann et al., 2004, Möllmann et al., 2005).

From David et al. (2007): "High abundances inducing the appearance of a second period of high zooplankton production in the estuary of Gironde, France, may impact on the trophic food web during a period previously characterized by low abundances of zooplanktonic preys (Bollens et al., 2002), particularly its utilization by fish and juveniles of shrimps that develop in this area (Sorbe, 1981) since copepods can be important components of the diet of bentho-pelagic invertebrates (Turner, 2004)."

From Katsanevakis et al. (2014): "Due to its high abundances and grazing abilities, *A. tonsa* can change energy and matter flows between pelagic and benthic systems and modify trophic structure of invaded ecosystems (Leppäkoski et al., 2002). *Acartia tonsa* can serve as a biological control of algal blooms (which would be considered as having a positive impact on water purification since it is a species that controls harmful species that have a negative impact on water quality). "

Replacement of *A. clausi* by *A. tonsa* appears detrimental for the next trophic level in European estuaries (Werbrouck et al., 2016).

Q13- Associated diseases, parasites, and travellers (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests A*cartia tonsa* would have high impacts in many areas associated with diseases, parasites or travelers in the RA area. There is a great deal of general information on impacts from parasites and epibionts for copepods, including documented impacts for the same genus. There is documented low host specificity and a known host species, *Centropages hamatus*, plus a number of closely related species that occur in the RA area. Information has been extrapolated from taxa that are closely related to *A. tonsa*.

Background Information: From Rawlings et al. (2007): "Huq et al. (2005) demonstrated a significant correlation between zooplankton blooms and cholera cases in Bangladesh. Copepods appear to be especially important because they are the most numerous and widely dispersed zooplankton with which the incidence and distribution of *Vibrio cholerae* have been associated (Huq et al., 1984, Huq et al., 1986, Tamplin et al., 1990) and because they possess a chitinous exoskeleton that is attractive to vibrios. Whether or not *V. cholerae* is a symbiont of plankton has yet to be shown. However, *V. cholerae* does occur in the gut and on the surface of zooplankton, such as *A. tonsa.* The availability of new surfaces provided by nauplii and eggs may be especially important to the population dynamics of *V. cholerae*."

From Bielecka and Boehnke (2014): "Epibiosis and parasitism are widespread in the zooplankton communities of marine and brackish environments (Hirche, 1974, Ho and Perkins, 1985, Timofeev, 1997, Hu and Song, 2001, Visse, 2007). Epibiotic overgrowth and parasitic infestation most often affect pelagic Copepoda (Wiktor and Krajewska-Sołtys, 1994, Timofeev, 2002, Visse, 2007, Walkusz and Rolbiecki, 2007). Both parasitism and epibiosis are considered harmful to planktonic animals. Overgrowths of epizoic Protozoa can reduce swimming speed in Copepoda, especially when the antennae are heavily infested. Heavily-infested specimens are also more visible to predators, becoming easy prey for planktivorous animals (Chiavelli et al., 1993, Visse, 2007). Kimmerer and McKinnon (1990) described cases of *Paracalanus indicus* infested with parasitic Dinoflagellata (*Atelodinium* sp.) in the Indian Ocean. They reported that dinoflagellates formed a plasmodium that wrapped around the host's body, leading to its death. Other authors examined the effect of the parasite *Ellobiopsis* sp. On the fecundity of *Calanus helgolandicus* in the Bay of Biscay. Parasitism by *Ellobiopsis* sp. Has the potential to reduce the fecundity of copepods: a reduction in size of both the seminal vesicle and the developing spermatophore sac was noted in parasitized males of *Calanus helgolandicus* (Albaina and Irigoien, 2006).The occurrence of epizoic filter-feeding Protozoa (*Vorticella* and *Zoothamnium*) and parasitic Protozoa (*Ellobiopsis*) on Calanoida was noticed in the Gulf of Gdańsk in 1998, 1999 and 2006. The relatively high (4–16% of all calanoids) level of infestation varied depending on the type of infestation (0.1–13% of the population of particular taxa). The dominant copepods – *Acartia* spp., *Temora longicornis* and *Centropages hamatus* – were attacked the most frequently (from 10.5% to 54% of all infested calanoids). Epibiosis and parasitism were observed on all copepod developmental stages (adults, juveniles and nauplii). Epibionts and parasites were located on different parts of the body, but mainly on the prosome. Infestation by epibionts and parasites was not restricted to calanoid copepods: it was also detected in non-negligible numbers on other crustaceans, namely, Harpacticoida, Cladocera (*Bosmina* sp.) and Cirripedia larvae (nauplii) in the Gulf of Gdańsk."

High levels of PSP toxin can be accumulated in copepod grazers such as *Acartia tonsa*, supporting the hypothesis that zooplankton may serve as PSP toxin vectors to higher trophic levels (Teegarden and Cembella, 1996).

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Acartia tonsa* would have high genetic impact in few areas or moderate genetic impact in many areas on other species in the RA area. There are co-occurring native copepods of the same genus in the RA area, but it seems doubtful *A. tonsa* would pose a risk for hybridizing with them (i.e. clades within the species do not seem to hybridize, suggesting that chances of species within the genus doing so may be that much lower).

Background Information: There is no specific information found. *Acartia bifilosa*, *Acartia clausi*, and *Acartia longiremis* have been reported from Hudson Bay, the Nelson River, and the brackish Nelson Estuary (Rochet and Grainger, 1988, Harvey et al., 2001, Stewart and Lockhart, 2005) North/South Consultants Inc. 2006). Although there is no information for *A. tonsa*, Goetze (2008) found that *Centropages typicus* females experience the highest incidence of heterospecific mating interactions with *Centropages hamatus*, and *Temora longicornis*, since they encounter heterospecific males at rates up to 100+ encounters/ female/day, ca., one order of magnitude higher than encounter rates with conspecifics – they suggested that heterospecific mating attempts may be a common feature of the reproductive ecology of planktonic copepods and may incur substantial fitness costs to the individuals involved.

Chen and Hare (2008) found that different clades of *Acartia tonsa* do not hybridize in Chesapeake Bay, suggesting that this may indicate important ecological divergence in marine holoplankton and a "hidden" biodiversity.

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Acartia tonsa* would have high impacts in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be moderately impacted given the moderate effects of this copepod on ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There were no studies found, but there are no invertebrate or plant species at risk in Hudson Bay. Given that bowhead whales (*Balaena mysticetus*) are plankton feeders, any impacts on plankton communities may impact this species. The probability or magnitude of such cascading effects is unknown. Likewise, potential impacts on larval species of at-risk fish are undetermined. As its invasion could bring cascading effects at higher trophic levels, species of fish at-risk including: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and the thorny skate, *Amblyraja radiata* could be affected.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Acartia tonsa* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. There is information on the ecological role of this species as a source of prey for fish from which impacts of this species to fish populations, communities and ecosystems in the RA can be extrapolated.

Background Information: *Acartia* spp. (including *A. tonsa*) constitute significant prey for fish in the Baltic (Möllmann et al., 2004, Möllmann et al., 2005). A positive impact is that *A. tonsa* has been used to produce live feed for aquaculture organisms (Sørensen et al., 2007), e.g. turbot rearing in the Black Sea. Its prevailing distribution in areas devoted to intense aquaculture in the Ponto-Mediterranean region, rather than in harbor areas, suggests an introduction associated with aquaculture (Belmonte and Potenza, 2001).

"They are an important food source for many commercial fish species (Sullivan et al., 2007). Several studies indicate they aggregate near the ocean floor during the day and rise closer to the surface at night. This behavior likely helps *A. tonsa* avoid predators who rely on vision to locate and capture prey (Fulton, 1984)." (https://en.wikipedia.org/wiki/Acartia_tonsa)

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Acartia tonsa* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: *Acartia tonsa* is a cryptic species with an almost worldwide distribution in coastal subtropical and temperate waters (https://www.nobanis.org/globalassets/speciesinfo/a/acartia-tonsa/acartia-tonsa.pdf). Ballast water seems to be the most likely means of transferring this species (Gubanova, 2000). It is considered established in the entire Baltic Sea (Pienimäki and Leppäkoski, 2004, Telesh, 2008) and reported from the Black, Caspian and Mediterranean seas (Kurashova and Abdullayeva, 1984, Gaudy and Viñas, 1985, Gubanova, 2000). Recent genetic studies point at the East coast of the USA as being the donor region for European *A. tonsa* (Drillet et al., 2008).

Centropages typicus Krøyer, 1849

Phylum: Arthropoda
Class: Hexanauplia
Order: Calanoida
Family: Centropagidae

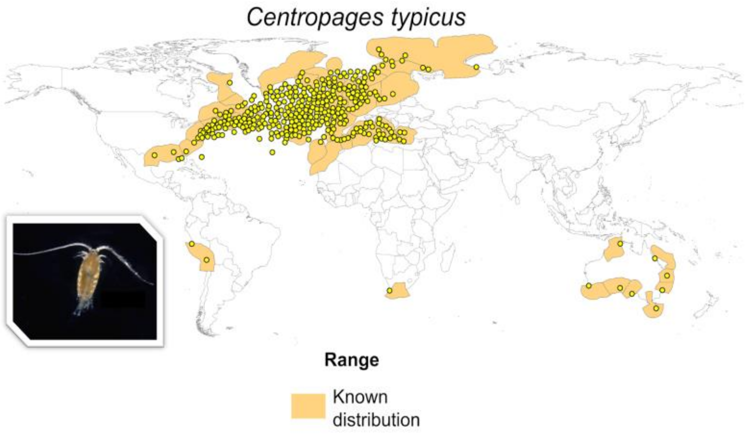

Figure 28: Ecoregions where Centropages typicus is distributed (orange). These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. It is not invasive anywhere in the world, but it has traits related to invasiveness Centropages typicus occurrence points were harvested from OBIS (https://obis.org/). Picture of C. typicus modified from https://zooplankton.no/.

CMIST scores for *C. typicus:*

Mean adjusted Likelihood of Invasion: 2.65

Mean adjusted Impact of Invasion: 1.93

Mean adjusted Overall CMIST score: 5.13

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Centropages typicus* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: There were no reports of *C. typicus* found in Hudson Bay (Stewart and Lockhart, 2005, Brown et al., 2016, Chain et al., 2016, Dispas, 2019). Regions marked in orange are regions where according to OBIS, there are registers of this species and that we highlight as outside of their described normal distribution (Figure 28).

Q2- Rate of introduction (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Centropages typicus* could arrive frequently in high numbers in the RA area. The species is present in connected ports and there is clear evidence of arrival from multiple studies and vectors of transport.

Background Information: Ships visiting Churchill can carry *Centropages* spp. on their hulls (1 of 13 ships tested; carried 563 individuals) and/or *C. typicus* in their ballast water (1 of 32 ships tested; 1,892,957 individuals) (Chan et al., 2015). Five individuals were found in the fouling community of one ship returning from an Arctic voyage (Chan et al., 2016). Exposure occurs during the open water shipping season. Ware et al. (2016) found *C. typicus* in the ballast water of 3 of 8 vessels that arrived in Svalbard, (mean = 19.7 +- 31.9 SE individuals m^3, present in 44% of samples, 37.5% of ships). This species is distributed throughout the north and east Atlantic and from Newfoundland to Chesapeake Bay along the North American coast (https://www.gbif.org)

Q3- Survival (suitable habitat) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable habitat for *Centropages typicus.* The species is present in coastal, estuarine and shallow waters. There is reliable information about seabed morphological characteristics in the National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: This species is a temperate calanoid copepod with a wide range of distribution in the North Atlantic and adjacent shelf seas (Carlotti and Harris, 2007; and references therein). It has neritic-coastal distribution, being confined to inner regions of shelves or to shallow banks (Durbin and Kane, 2007), and it can also be found in shallow estuaries and embayments in areas with high salinity (Carlotti and Harris, 2007). It occurs in oceanic regions near continental slopes (Beaugrand et al., 2007). It cannot withstand more than a few days without food due to low lipid reserves, explaining its lower observed abundance in open-ocean waters (Carlotti and Harris, 2007). Highest abundances are observed in water depths of <59 m in inshore shelf regions and decreased abundances towards their outer edges (Kane, 1997). *Centropages typicus* prefers warmer waters and when a thermocline is present will remain in the surface layer (Durbin and Kane, 2007).

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable anthropogenic conditions for *Centropages typicus.* Minimum temperature tolerance is -2°C. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Chan et al. (2015) described *C. typicus* as an Arctic species that could tolerate environmental conditions in Churchill (although no other studies designates its distribution as an Arctic species). It is present over a wide range of temperatures, ranging from 1 to 24°C (Bonnet et al., 2007). Its abundance increases with temperatures between 13 and 20°C. Temperature appears to be the key factor explaining the variability in its spatial and temporal distribution on shelves where feeding is not food-limited, whereas offshore distribution is probably food-limited (Carlotti and Harris, 2007). *Centropages typicus* finds its physiological optimum around 15–18°C (Halsband-Lenk et al., 2002). Although there are other studies that found the optimal temperature tolerance lower: ranging from - 2.0 to 29.2°C with greatest abundances at temperatures between 9 and 25°C (Kane, 1997). Sell et al. (2001) found increased feeding rates at 6°C, but only two temperatures were tested in the study (6 and 11°C). It occurs over a wide range of Chl *a* concentrations, but mainly below 6 mg Chl *a* m^3 (Bonnet et al., 2007).

Q5- Establishment (reproductive requirements) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are almost always available for *Centropages typicus* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Given that the species has been shown to reproduce at low temperatures (2, 4, and 8°C depending on population), a large proportion of the RA would reach summer temperatures at or above these thresholds so would be expected to be suitable for reproduction. The species requirements are well demonstrated and there are reliable environmental data layers for the RA area to show a match.

Background Information: Temperature is the controlling factor for reproduction and development. It is potentially very fecund between 10 and 24°C with an optimal temperature around 20°C (Carlotti and Harris, 2007). Halsband-Lenk et al. (2002) found that development times decreased with increasing temperatures; eggs hatched within 2 days from 15 to 25°C, and within 5 days at lower (5°C) temperatures. Low temperatures apparently inhibited reproductive activity off Helgoland (North Sea) where winter temperatures are sometimes below 0°C (Halsband-Lenk and Hirche, 2001). Bigelow (1926) suggested a minimum breeding temperature of 8–12°C, however, it has been registered in minimum winter temperatures about 4°C (Kane, 1997). *Centropages typicus* from the North Sea were able to produce eggs at 2°C, whereas the Mediterranean population did not (Halsband-Lenk et al., 2002). Neither resting eggs, nor over-wintering stages have been identified in *C. typicus*. Survival of robust adults in inner parts of shelves and continuous spawning of females to initiate new cohorts during conditions of good ‘‘environmental windows’’ might be a strategy to maintain the populations (Carlotti and Harris, 2007). The natural variety of prey is important for maximizing the fitness of *C. typicus* (Ianora et al., 2007). Larval development is rapid, with adulthood being reached after 16–49 days, depending on temperature and food conditions contributing to the success of this species. (Ianora et al., 2007; and references therein).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Centropages typicus.* There is some reliable information, but only generalities, and no species-specific information on predation that would allow for extrapolation to the RA area. Also, there is no information on whether or not the parasites/diseases would be present in the RA area or transported with the host species in vectors.

Background Information: Major predators of *C. typicus* are Chaetognaths together with Gammaridae, Hyperiidae and Euphausiids, and intraspecific predation (cannibalism) can also regulate the population (Stegert et al., 2012). *Centropages typicus* can be infested with bacteria that invades the entire body cavity of their hosts causing behavioral anomalies in swimming mode but normal internal organs and oogonal development (Ianora et al., 1990). This may not act directly regulating the population, but it could affect performance. Generally speaking, at the individual level, parasitism can delay development, induce intersexuality, and cause sterility and even death in the host. The effect at the community level will depend on the incidence of infection in natural populations (Ianora et al., 1990; and references therein). They can be infested with extracellular parasitic infestations due to the dinoflagellate *Blastodinium.* Some studies show that this can affect the reproductive biology of *C. typicus* (Jepps, 1937, Cachon et al., 1987) while some others found no effect (Ianora et al., 1990).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Centropages typicus* has the capacity for a wide range of natural dispersal in the RA area.

Background Information: It is quite likely that this species will not prosper in oceanic waters considering its low tolerance to starvation (Dagg, 1977). It is a broadcast spawner (eggs and larvae can be dispersed) (Ianora et al., 2007). In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Centropages typicus* has the capacity for a wide range of anthropogenic dispersal in the RA area. There seems to be a great deal of solid evidence for wide range dispersal through uptake in ballast and biofouling on vessels moving within the RA area.

Background Information: *C. typicus* could be transported by hull fouling and ballast water (Chan et al., 2015, 2016). Ballast water and hull fouling transported by domestic coastal resupply and other boat traffic, and possibly hull fouling, could spread *C. typicus* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Centropages typicus* would have high impacts in few areas or moderate impact in many areas on population growth in the RA area. There is a moderate amount of reliable information on the ecological role of this species as a predator from which impacts of this species to populations in the RA area can be extrapolated.

Background Information: There are no studies found for *C. typicus* as an invasive species causing an ecological impact. Nevertheless, it can be inferred by its ecological role that it could have an impact on population growth if it invades a new region given that *C. typicus* is mostly carnivorous or omnivorous, feeding on a wide spectrum of prey. This includes phytoplankton, from small 3–4 lm equivalent spherical diameter to large diatoms, and animal prey such as ciliates, appendicularia, eggs and nauplii of copepods, up to yolk-sac fish larvae larger than 3 mm length (Calbet et al., 2007).

From Blanco-Bercial and Bucklin (2016): "The projected increase in *C. typicus* populations linked to increasing temperatures (Stegert et al., 2012) could have a significant impact on ecosystems and fisheries, especially if combined with a decrease in the abundances of other copepod species, such as *Pseudocalanus* spp. (Erikson et al., 2014, Kane, 2014) which are the preferred prey of these economically important fish species (Petrik et al., 2009). *Centropages typicus* is not, however, a preferred prey item of haddock and cod larvae (Petrik et al., 2009)."

Q10- Impact on communities **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests *Centropages typicus* would have high impacts in few areas or moderate impacts in many areas on communities in the RA area. There are moderate amounts of reliable information on the ecological role of this species as a predator from which impacts of this species to communities in the RA area can be extrapolated.

Background Information: There are no studies found for *C. typicus* as an invasive species causing an ecological impact. Nevertheless, it can be inferred by its ecological role that it could have an impact in communities if it invades a new region given that *C. typicus* is mostly carnivorous or omnivorous, feeding on a wide spectrum of prey. This includes phytoplankton, from small 3–4 lm equivalent spherical diameter to large diatoms, and animal prey such as ciliates, appendicularia, eggs and nauplii of copepods, up to yolk-sac fish larvae larger than 3 mm length (Calbet et al., 2007).

From Blanco-Bercial and Bucklin (2016): "The projected increase in *C. typicus* populations linked to increasing temperatures (Stegert et al., 2012) could have a significant impact on ecosystems and fisheries, especially if combined with a decrease in the abundances of other copepod species, such as *Pseudocalanus* spp. (Erikson et al., 2014, Kane, 2014), which are the preferred prey of these economically important fish species (Petrik et al., 2009). *Centropages typicus* is not, however, a preferred prey item of haddock and cod larvae (Petrik et al., 2009)."

Q11- Impact on habitats (Score = 1, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Centropages typicus* would have low or no impact on habitat in the RA area. There is a lack of information demonstrating habitat impacts. Also, this species likely does not affect the physical habitat, only the organisms occupying it. Since it is chiefly a predator, it is affecting mostly zooplankton species, and likely not impacting water properties.

Background Information: There are no studies found for *C. typicus* as an invasive species causing an ecological impact on habitats.

Q12- Impact on ecosystem function (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Centropages typicus* would have high impact in few areas or moderate impact in many areas on ecosystem function in the RA area. There is reliable information on the ecological role of species as a predator from which impacts of this species to populations, communities and ecosystems in the RA area can be extrapolated.

Background Information: There are no studies found for *C. typicus* as an invasive species causing an ecological impact. Nevertheless, it can be inferred by its ecological role that it could have an impact on ecosystem function if it invades a new region given that: Copepods in general are abundant throughout world oceans and serve as a major food source for higher trophic levels, including larval and juvenile fish species (Stegert et al., 2012; and references therein). *Centropages typicus* is an important species in the mesozooplankton assemblage of the fluctuating neritic coastal marine environment (Carlotti and Harris, 2007; and references therein). Its role in pelagic food webs needs to be evaluated further, given the contribution of the species to zooplankton biomass. It is unclear whether it has a strong grazing impact on the first trophic levels (Calbet et al., 2007) perhaps because of the diversity of its prey. However, it probably is an important link between primary producers and fish larvae (Carlotti and Harris, 2007). Changes in zooplankton modes (increasing abundance of *C. typicus* in the ecosystem) may reflect broad changes in ecosystems and in fish stocks such as seen in Pershing et al., (2005).

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Centropages typicus* would have high impacts in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. There is a great deal of general information on impacts from parasites and epibionts for copepods, including documented impacts for the same genus. There is documented low host specificity and a known host species, *Centropages hamatus*, plus a number of closely related species that occur in the RA area. This information is extrapolated from taxa that are closely related to *C. typicus*.

Background Information: No studies were found that are specific to *Centropages typicus*. However Bielecka and Boehnke (2014) studied epibionts and parasites of several copepods and stated that: "Epibiosis and parasitism are widespread in the zooplankton communities of marine and brackish environments (Hirche, 1974, Ho and Perkins, 1985, Timofeev, 1997, Hu and Song, 2001, Visse, 2007). Epibiotic overgrowth and parasitic infestation most often affect pelagic Copepoda (Wiktor and Krajewska-Sołtys, 1994, Timofeev, 2002, Visse, 2007, Walkusz and Rolbiecki, 2007). Both parasitism and epibiosis are considered harmful to planktonic animals. Overgrowths of epizoic Protozoa can reduce swimming speed in Copepoda, especially when the antennae are heavily infested. Heavily-infested specimens are also more visible to predators, becoming easy prey for planktivorous animals (Chiavelli et al., 1993, Visse, 2007). Kimmerer and McKinnon (1990) described cases of *Paracalanus indicus* infested with parasitic Dinoflagellata (*Atelodinium* sp.) in the Indian Ocean. They reported that dinoflagellates formed a plasmodium that wrapped around the host's body, leading to its death. Other authors examined the effect of the parasite *Ellobiopsis* spp. on the fecundity of *Calanus helgolandicus* in the Bay of Biscay. Parasitism by *Ellobiopsis* spp. has the potential to reduce the fecundity of copepods: a reduction in size of both the seminal vesicle and the developing spermatophore sac was noted in parasitized males of *C. helgolandicus* (Albaina and Irigoien, 2006). The occurrence of epizoic filter-feeding Protozoa (*Vorticella* and *Zoothamnium*) and parasitic Protozoa (*Ellobiopsis*) on Calanoida was noticed in the Gulf of Gdańsk in 1998, 1999 and 2006. The relatively high (4–16% of all calanoids) level of infestation varied depending on the type of infestation (0.1–13% of the population of particular taxa). The dominant copepods – *Acartia* spp., *Temora longicornis* and *Centropages hamatus* – were attacked the most frequently (from 10.5% to 54% of all infested calanoids). Epibiosis and parasitism were observed on all copepod developmental stages (adults, juveniles and nauplii). Epibionts and parasites were located on different parts of the body, but mainly on the prosome. Infestation by epibionts and parasites was not restricted to calanoid copepods: it was also detected in non-negligible numbers on other crustaceans, namely, Harpacticoida, Cladocera (*Bosmina* sp.) and Cirripedia larvae (nauplii) in the Gulf of Gdańsk."

Q14- Genetic impact (Score = 3, Certainty = 1)

Score Rationale: Little to no reliable information suggests *Centropages typicus* would have high genetic impact in many areas on other species in the RA area. Species demonstrated experimentally to mate with a species found within the RA area (*Centropages hamatus*), although it is unknown whether these matings can produce viable offspring.

Background Information: No studies were found, but *C. hamatus* occurs in the Churchill River estuary and elsewhere in Hudson Bay and Hudson Strait (Harvey et al., 2001) North/South Consultants Inc. 2006). In the North Sea, *C. typicus* and *C. hamatus* coexist in a seasonal succession (*C. typicus* prefers the saline and relatively warm waters of Atlantic origin and is usually abundant in the second half of the year) (Bonnet et al., 2007; and references therein). *Centropages hamatus* is a cold water species while *C. typicus* shows a wide tolerance for temperature (Bonnet et al., 2007). Goetze (2008) found that *C. typicus* females experience the highest incidence of heterospecific mating interactions with *Centropages hamatus*, and *Temora longicornis*, since they encounter heterospecific males at rates up to 100+ encounters/female/day, ca. one order of magnitude higher than encounter rates with conspecifics - they suggested that heterospecific mating attempts may be a common feature of the reproductive ecology of planktonic copepods and may incur substantial fitness costs to the individuals involved.

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Centropages typicus* would have high impact in few areas or moderate impact in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be moderately impacted given the moderate effects of this copepod on ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: No studies were found, but there are no invertebrate or plant species at risk in Hudson Bay. Given that bowhead whales (*Balaena mysticetus*) are plankton feeders, any impacts on plankton communities may impact this species. The probability and magnitude of such cascading effects is unknown. Likewise, the potential impacts on larval species of at risk fish is undetermined. As its invasion could bring cascading effects at higher trophic levels, species of fish at risk include: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and thorny skate, *Amblyraja radiata* could be affected.

Q16- Impact on fisheries (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Centropages typicus* would have high impacts in few areas or moderate impacts in many areas on fished species in the RA area. Reliable information on the ecological role of the species as a plankton predator from which impacts of this species to populations, communities and ecosystems in the RA area can be extrapolated.

Background Information: *Centropages typicus* is considered to be an important food source for many larval stages of commercially important fish stocks such as anchovy in European waters (Carlotti and Harris, 2007). Zooplanktivorous fish used for subsistence could be affected.

From Blanco-Bercial and Bucklin (2016): "The projected increase in *C. typicus* populations linked to increasing temperatures (Stegert et al., 2012) could have a significant impact on ecosystems and fisheries, especially if combined with a decrease in the abundances of other copepod species, such as *Pseudocalanus* spp. (Erikson et al., 2014, Kane, 2014), which are the preferred prey of these economically important fish species (Petrik et al., 2009). *Centropages typicus* is not, however, a preferred prey item of haddock and cod larvae (Petrik et al., 2009)."

Q17- Past invasion history (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Centropages typicus* is not invasive anywhere in the world, but it has traits related to invasiveness. There is a moderate amount of information about the species in the literature, but none of it suggests it is invasive, only that it can be found in shipping pathways. However, there is a high likelihood that it could become invasive if introduced since it has traits related to invasiveness: rapid maturation, high fecundity, broadcast spawning and a pelagic life history which would allow for rapid spread.

Background Information: There are no studies that have detected this species as an invasive one, although there are studies such as the ones from Chan et al. (2015, 2016) and Ware et al. (2016) that name the species as potential non-native species to the Arctic region. Howland (unpubl. data) found a specimen in ballast water. In its native range the major centers of distribution for the species in the eastern North Atlantic are the North Sea, English Channel, Celtic Sea and Bay of Biscay (Beaugrand et al., 2007). The species is dominant on the Scotian Shelf and present year round, but it has not been reported in the Gulf of St Lawrence (Durbin and Kane, 2007; and references therein). Although, *C. typicus* has been considered a rare or sporadic immigrant in the North Sea or was not mentioned at all in earlier studies, whether this increase is exceptional, induced by a singular inflow event, or a long-term trend, remains to be determined (Bonnet et al., 2007; and references therein).

Eurytemora affinis affinis (Poppe, 1880)

- Phylum: Arthropoda
  - Class: Hexanauplia
    - Order: Calanoida
      - Family: Temoridae

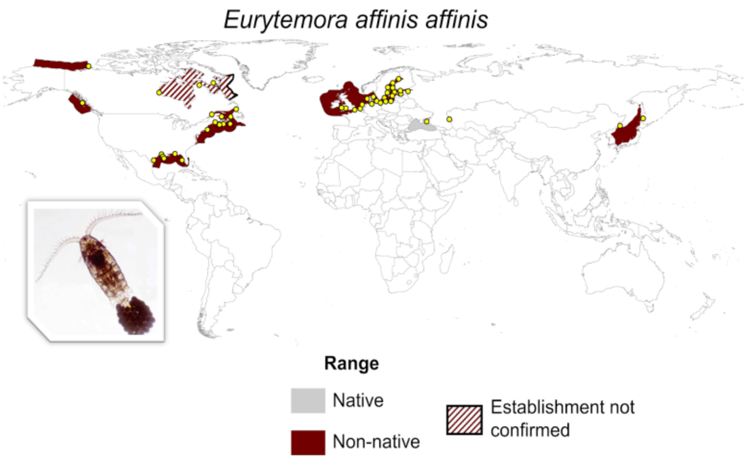

Figure 29: Ecoregions where Eurytemora affinis affinis is distributed (orange). Consider that E. affinis is composed of clades that are morphologically indistinguishable and that it can bring some confusion in identifying its status where it is distributed. For simplicity reasons, native range (grey) has been identified while non-native range (red) should be carefully considered (see Q14 and Q17). These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Eurytemora affinis affinis occurrence points were harvested from OBIS (https://obis.org/). Note that points in the Eastern Canadian Arctic were mainly detected through metabarcoding (Chain et al., 2016) and only individuals of E. affinis were found in Churchill (Dispas, 2019), yet establishment has not been confirmed (red stripes). Picture of E. affinis modified from https://www.hgsc.bcm.edu/arthropods/eurytemora-affinis-genome-project (Carol Lee)

CMIST scores for *E. affinis:*

Mean adjusted Likelihood of Invasion: 2.71

Mean adjusted Impact of Invasion: 1.98

Mean adjusted Overall CMIST score: 5.38

Q1- Present status in the area (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* is established in the RA area. It has been found at more than one location and time in the RA area and there is a logical source through shipping. However, certainty is moderate since there is the possibility of taxonomic uncertainty based on misidentification in metabarcoding.

Background Information: This species has been identified in the brackish waters of the Nelson River estuary (Baker, 1989, Zrum, 2000) and was found in the Churchill port in 2014 (Dispas, 2019). Its presence has been identified by metabarcoding in Churchill (44027 reads), Deception Bay (2141 reads) and Iqaluit (174 reads) (Chain et al., 2016).

Q2- Rate of introduction (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* could arrive frequently in high numbers in the RA area. The species has been found on arriving vessels and is present in a number of connected port ecoregions.

Background Information: This species is dominant in the ballast of vessels arriving in Deception Bay and Churchill (Tremblay, 2017, Dispas, 2019). *Eurytemora affinis* is present in the St-Lawrence estuary, and it could be transported in the ballast water tanks of the MV *Arctic*, which regularly transits from Quebec City to Raglan Mines in Deception Bay. Deception Bay normally receives the highest average annual domestic arrivals (Goldsmit et al., 2019). The species is established in freshwater systems (Great Lakes and Nelson River) with connections to Hudson Bay (Zrum, 2000).

Q3- Survival (suitable habitat) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable habitat for *Eurytemora affinis.* It can live in a wide range of habitats which occur through much of the RA area. There is reliable information in National Center for Ecological Analysis and Synthesis ecosystem types data layers to support classification.

Background Information: The species complex *Eurytemora affinis* has a geographic distribution from subtropical to subarctic regions that spans North America, Asia, and Europe, with habitat types that range from hypersaline salt marshes and brackish estuaries to completely fresh water (Saunders, 1993, Lee, 1999). It is a strongly euryhaline and eurythermal calanoid that tolerates high turbidity, and is widely distributed in estuaries, brackish seas and salt marshes in the Northern Hemisphere (Andersen and Nielsen, 1997). In the Baltic Sea *E. affinis* is a dominant copepod, both in littoral and pelagic ecosystems (Telesh and Heerkloss, 2004). In Asia, it is known from the Caspian Sea and from fresh water lakes in Japan (Lee, 2000, Dussart and Defaye, 2002). It is found to have a voracious appetite when residing in freshwater habitats (Lee et al., 2013). The species has been known to habitat partition and genetically subdivide (genetically distinct clades can show a pattern of niche partitioning) (Winkler et al., 2008).

Q4- Survival (suitable environment) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests most of the RA area offers suitable environmental conditions for *Eurytemora affinis.* The species can survive in other locations at sub-zero temperatures and tends to be cold-adapted. Even though there are reliable environmental data layers for the RA area to show a match, there is some uncertainty of the RA areas suitability due to high levels of variability in physiological tolerance and species requirements among populations.

Background Information: High-food concentration increases low-salinity tolerance, allowing saline populations to invade freshwater habitats (Lee et al., 2013, Hammock et al., 2016, Lee, 2016). While *E. affinis* has among the broadest salinity ranges of invertebrates, individual *E. affinis* populations are unable to survive the entire salinity range spanning saline to freshwater habitats. An evolutionary shift at the population level is required for the physiological shift to occur, enabling the once saline populations to persist in freshwater habitats (Lee, 2016). Each environment has influenced individual physiological performance and ultimately the ecological performance of each population (e.g. populations from St-Lawrence respond differently to salinity changes when compared to populations from France (Beyrend-Dur et al., 2009)). Maximum temperature will limit *E. affinis*, though independently of food concentration (Hammock et al., 2016). Kinne (1964) found that resistance to temperature extremes increased with salinity. Diapausing eggs of *E. affinis* can survive in the sediment at least for several decades (Katajisto, 1996) and become injected into the population at a later time (Lee, 2016). It possesses ion regulatory leg organs for ion uptake in their swimming legs (Gerber et al., 2016) (Johnson et al. 2014; from Lee 2016). *Eurytemora* species have a tendency toward cold adaptation, such that species ranges beyond cold regions might be limited by warmer temperature (Katona, 1970). Models done by Ware et al. (2016) projected that *E. affinis* could survive in HB in the future. *Eurytemora affinis* is present in the St-Lawrence estuary as well as in the Nelson River estuary (Baker, 1989, Zrum, 2000, Lee, 2016), both of which experience extended periods with sub-zero temperatures (-1.5°C) similar to what would be expected under ice in the assessment region for much of the winter.

Q5- Establishment (reproductive requirements) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests reproductive requirements are almost always available for *Eurytemora affinis* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >= 2°C (minimum reproductive temperature) together with its known wide salinity tolerance, resulted in selecting the majority of the RA area that would be suitable for reproduction at some point during the open water season. There is good experimental data on reproductive requirements and environmental layers to evaluate the extent of the RA area that would have suitable conditions for reproduction.

Background Information: *Eurytemora affinis* is recognized as a sibling species complex, marked by morphological stasis, high genetic divergence among clades and by reproductive isolation between several nearby populations, representing divergent evolutionary histories (Lee, 2000, Lee and Frost, 2002). Devreker et al. (2009) found that a combination of low temperature and high salinities resulted in negative impacts on reproductive parameters for *E. affinis*. Their minimum reproductive temperature is 5.5°C (Heinle and Flemer, 1975). Reproductive salinity is between 0 and 27 ppt (Lee and Peterson, 2003) but Katona (1970) demonstrated experimentally that *E. affinis* reproduces in salinity levels between 5 and 33 % and at temperatures between 2 and 23.5°C. The species has desiccation-resistant resting eggs, and diapausing eggs can survive in the sediment at least for several decades (Katajisto, 1996) and become injected into the population at a later time as a means of escaping stressful conditions (Lee and Bell, 1999; and references therein). Knowing this and that it could survive in HB in the future (modelling work by Ware et al. (2016)), this could increase the chances of reproducing on a long-term scale.

Q6- Establishment (natural control agents) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information is available that suggests natural control agents could slow population growth in the RA area for *Eurytemora affinis*. There are closely related species in the RA area (e.g., mysids and herring) that could act as predators, but no information to indicate they could severely restrict population growth. There is some information on predators for the species and zooplankton is well known as food for a variety of predators that would be expected in the RA area.

Background Information: *Eurytemora affinis* is a preferred prey for dominant zooplanktivores such as herring, sprat, mysids and predatory cladocerans (Lehtiniemi and Gorokhova, 2008; and references therein). Decreased food availability that can be caused by other invasive species such as the clam *Corbula amurensis*, can narrow the realized salinity niche of *E. affinis* (Hammock et al., 2016). Aaser et al. (1995) show that the mysid *Neomysis integer* and the stickleback *Gasterosteus aculeatus* may impact the species in the field and in enclosure experiments.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eurytemora affinis* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for widespread dispersal potential given diapausing eggs and planktonic larvae/adults.

Background Information: Under stressful conditions, diapausing eggs of *E. affinis* can survive in the sediment at least for several decades (Katajisto, 1996) and could be injected into the environment at a later time through disturbance of sediments via storm events and upwelling (Lee and Bell, 1999; and references therein).

Although the species is a weak swimmer (Lee, 2016), in the HB LME, alongshore currents (Granskog et al., 2011) could disperse various life stages of species (eggs, larvae, adults) counterclockwise around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is solid evidence of widespread dispersal through ballast and the species has been observed in the ballast of ships arriving in the RA area. There is some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: *Eurytemora affinis* is dominant in the ballast of vessels arriving in Deception Bay and Churchill (Howland et al unpubl.). Ballast transported by domestic coastal resupply and other boat traffic, could spread *E. affinis* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Recent invasions by *E. affinis* from saline estuaries and salt marshes into inland freshwater lakes and reservoirs have been mediated by human activity (Saunders, 1993, Lee, 1999, Winkler et al., 2008). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* would have high impacts in few areas or moderate impacts in many areas on population growth in the RA area. There is some evidence of impacts to protozoan ciliates and rotifer populations and the food web/ecosystem through grazing and it has a similar ecology to the more extensively studied *Acartia tonsa* which has been shown to have high impacts. Most of the information is from freshwater systems or inferred from other plankton species with similar ecologies.

Background Information: The ecological effects of invasive copepods in North America are not well understood. No native species have been eliminated by invasive copepods, although some native copepods have experienced reduced abundances or changes in their distribution in time and space after their introduction (Francis, 2012).

From GLANSIS: “*Eurytemora affinis* has the ability to feed on toxic cyanobacteria and dinoflagellates (*Dinophysis* spp.) (Engström et al., 2000, Setälä et al., 2009). While these do not appear to be their preferred food source, consumption of toxic phytoplankton results in the buildup of toxins in zooplankton tissue and feces, which consequently can accumulate in benthic organisms, fish, and organisms further up the food chain (Engström et al., 2000, Lehtiniemi et al., 2002, Setälä et al., 2009). Copepods are also common hosts for fish parasites (Piasecki et al., 2004). In particular, *E. affinis* is a probable host and vector for plerocercoids that can infect striped bass in the Sacramento-San Joaquin Estuary (Arnold and Yue, 1997).

Mesocosm experiments indicate that *E. affinis* has the potential to control some populations of protozoan ciliates and rotifers when these prey items are found at high densities (Merrell and Stoecker, 1998, Feike and Heerkloss, 2009). Because *E. affinis* has become an abundant grazer in parts of the Great Lakes, it is possible that it has had important impacts on the food web—both adverse and beneficial (Lee et al., 2007).

Eurytemora affinis could be a significant prey item for fish and other planktivores. Thorp and Casper (2003) demonstrated such potential in an enclosure experiment with yellow perch (*Perca flavescens*) in the St. Lawrence River; 99% of *E. affinis* disappeared from fish

enclosures, presumably due to predation."
(https://nas.er.usgs.gov/queries/GreatLakes/FactSheet.aspx?SpeciesID=178)

From Lee et al. (2007): "*Eurytemora affinis* is a major component of food webs in estuarine and salt marsh ecosystems in the Northern Hemisphere. Its native range spans salinities from brackish to hypersaline (up to 40 PSU in salt marshes). *Eurytemora affinis* is a dominant grazer in the St. Lawrence estuary of North America and has become abundant in the Great Lakes since the opening of the St. Lawrence Seaway ca. 1959. In addition to large impacts on food web structure, such inland invasions could have implications for disease transmission, as *E. affinis* is a disease vector and a major carrier of Cholera (Colwell, 2004, Piasecki et al., 2004)."

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* would have high impacts in few areas or moderate impacts in many areas on communities in the RA area. There is some evidence of impacts to protozoan ciliates and rotifer communities and the food web/ecosystem through grazing and it has a similar ecology to the more extensively studied *Acartia tonsa* which has been shown to have high impacts. Most of the information is from freshwater systems or inferred from other plankton species with similar ecologies.

Background Information: The ecological effects of invasive copepods in North America are not well understood. No native species have been eliminated by the invasive copepods, although some native copepods have experienced reduced abundances or changes in their distribution in time and space after their introduction (Francis, 2012).

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Outbreaks of cholera are sometimes correlated with copepods, which are common hosts of *Vibrio cholerae* (Colwell, 2004, Piasecki et al., 2004, Lee et al., 2007). Cholera outbreaks tend to be associated with algal blooms and the rapid increase in copepods that follows (Piasecki et al., 2004). *Eurytemora* spp. are known to host *V. cholerae* and are the most common of known copepod hosts in Chesapeake Bay, where this has been studied (Colwell, 2004).

Eurytemora affinis has the ability to consume cyanobacteria and other toxic algal blooms; studies in the Baltic Sea indicate that this is likely an important mechanism of the biomagnification of toxins in organisms of economic importance, such as shrimp and fish (Engström et al., 2000, Karjalainen et al., 2008, Setälä et al., 2009).

Eurytemora affinis could be a significant prey item for fish and other planktivores. Thorp and Casper (2003) demonstrated such potential in an enclosure experiment with yellow perch (*Perca flavescens*) in the St. Lawrence River; 99% of *E. affinis* disappeared from fish enclosures, presumably due to predation."
(https://nas.er.usgs.gov/queries/GreatLakes/FactSheet.aspx?SpeciesID=178)

From Lee et al. (2007): "*Eurytemora affinis* is a major component of food webs in estuarine and salt marsh ecosystems in the Northern Hemisphere. Its native range spans salinities from brackish to hypersaline (up to 40 PSU in salt marshes). *Eurytemora affinis* is a dominant grazer in the St. Lawrence estuary of North America and has become abundant in the Great Lakes since the opening of the St. Lawrence Seaway ca. 1959. In addition to large impacts on food web structure, such inland invasions could have implications for disease transmission, as *E. affinis* is a disease vector and a major carrier of Cholera (Colwell, 2004, Piasecki et al., 2004)."

Q11- Impact on habitats (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora affinis* would have high impacts in few areas or moderate impacts in many areas on habitat in the RA area. Information is lacking on habitat impacts, and the species does not seem to have a physical impact on the water column per se - only on the organisms within it. There is no evidence of increasing water clarity or other changes to water quality parameters, but the possibility exists.

Background Information: There are no studies suggesting direct habitat impacts, however there is a possibility of impacts on water clarity as is seen in other grazers.

Q12- Impact on ecosystem function (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* would have high impacts in few areas or moderate impacts in many areas on ecosystem function in the RA area. There is some evidence of impacts to protozoan ciliates and rotifer populations/communities and the food web/ecosystem through grazing and it has a similar ecology to the more extensively studied *Acartia tonsa* which has been shown to have high impacts. Most of the information is from freshwater systems or inferred from other plankton species with similar ecologies.

Background Information: *Eurytemora affinis* is an important component of the estuarine food web, including its role as a food source for higher trophic levels (Strasser et al., 2011; and references therein).

The ecological effects of invasive copepods in North America are not well understood. No native species have been eliminated, although some native copepods have experienced reduced abundances or changes in their distribution in time and space after their introduction (Francis, 2012).

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Mesocosm experiments indicate that *E. affinis* has the potential to control some populations of protozoan ciliates and rotifers when these prey items are found at high densities (Merrell and Stoecker, 1998, Feike and Heerkloss, 2009). Because *E. affinis* has become an abundant grazer in parts of the Great Lakes, it is possible that it has had important impacts on the food web—both adverse and beneficial (Lee et al., 2007).

Outbreaks of cholera are sometimes correlated with copepods, which are common hosts of *Vibrio cholerae* (Colwell, 2004, Piasecki et al., 2004, Lee et al., 2007). Cholera outbreaks tend to be associated with algal blooms and the rapid increase in copepods that follows (Piasecki et al., 2004). *Eurytemora* spp. are known to host *V. cholerae* and are the most common of known copepod hosts in the Chesapeake Bay, where this has been studied (Colwell, 2004).

Eurytemora affinis has the ability to consume cyanobacteria and other toxic algal blooms; studies in the Baltic Sea indicate that this is likely an important mechanism of the biomagnification of toxins in organisms of economic importance, such as shrimp and fish (Engström et al., 2000, Karjalainen et al., 2008, Setälä et al., 2009).

Eurytemora affinis could be a significant prey item for fish and other planktivores. Thorp and Casper (2003) demonstrated such potential in an enclosure experiment with yellow perch (*Perca flavescens*) in the St. Lawrence River; 99% of *E. affinis* disappeared from fish enclosures, presumably due to predation." (https://nas.er.usgs.gov/queries/GreatLakes/FactSheet.aspx?SpeciesID=178)

From Lee et al. (2007): "*Eurytemora affinis* is a major component of food webs in estuarine and salt marsh ecosystems in the Northern Hemisphere. Its native range spans salinities from brackish to hypersaline (up to 40 PSU in salt marshes). *Eurytemora affinis* is a dominant grazer in the St. Lawrence estuary of North America and has become abundant in the Great Lakes since the opening of the St. Lawrence Seaway ca. 1959."

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora affinis* would have high impacts in few areas or moderate impacts in many areas associated with diseases, parasites or travelers in the RA area. Much general information on epibionts and parasites of copepods, which indicates a relatively broad suite of taxa can be transported on a wide range of copepod taxa, including ones found in the RA area.

Background Information: *Eurytemora affinis* is a major host of many pathogens, including *Vibrio cholerae*, *Vibrio vulnificus*, and *Vibrio parahaemolyticus* (Huq et al., 1983, Heidelberg et al., 2002, Colwell, 2004). Huq et al. (2005) demonstrated a significant correlation between zooplankton blooms and cholera cases in Bangladesh. *Vibrio cholerae* occurs in the gut and on the surface of zooplankter such as *E. affinis* (Rawlings et al., 2007). Hatching nauplii can affect *V. cholerae* attachment to both adults and eggs. The presence of hatched nauplii resulted in eggs with greater numbers of *V. cholerae* (Rawlings et al., 2007). *Eurytemora affinis* is inhabited by an enormous consortium of microbiota. Interestingly, there is a dramatic shift in the copepod microbiome during invasions, with parallel shifts in the microbiome composition during saline to freshwater invasions (Gelembiuk, 2015). In lab conditions, *E. affinis* fed actively on the toxic *Nodularia spumigena* even when other food items were present, with no significant effect on the survival, feeding and reproduction (Kozlowsky-Suzuki et al., 2003). *Eurytemora affinis* would be capable of degrading/detoxifying toxins, but with some delay after ingestion (Kozlowsky-Suzuki et al., 2003). This process could vary depending on the conditions (e.g. starvation or transfer to a non-toxic food source) to which the animals are subjected after exposure to a toxic diet (Svensson, 2000). Only a very small fraction of the calculated ingested toxin could be found in the animals. Thus, even though these grazers might act as a link transferring toxins to higher trophic levels, the relative importance of this indirect pathway seems limited (Kozlowsky-Suzuki et al., 2003).

Bielecka and Boehnke (2014) did a study on epibionts and parasites of several copepods and stated that : "Epibiosis and parasitism are widespread in the zooplankton communities of marine and brackish environments (Hirche, 1974, Ho and Perkins, 1985, Timofeev, 1997, Hu and Song, 2001, Visse, 2007). Epibiotic overgrowth and parasitic infestation most often affect pelagic Copepoda (Wiktor and Krajewska-Sołtys, 1994, Timofeev, 2002, Visse, 2007, Walkusz and Rolbiecki, 2007). Both parasitism and epibiosis are considered harmful to planktonic animals. Overgrowths of epizoic Protozoa can reduce swimming speed in Copepoda, especially when the antennae are heavily infested. Heavily-infested specimens are also more visible to predators, becoming easy prey for planktivorous animals (Chiavelli et al., 1993, Visse, 2007). Kimmerer and McKinnon (1990) described cases of *Paracalanus indicus* infested with parasitic Dinoflagellata (*Atelodinium* sp.) in the Indian Ocean. They reported that dinoflagellates formed a plasmodium that wrapped around the host's body, leading to its death. Other authors examined the effect of the parasite *Ellobiopsis* sp. on the fecundity of *Calanus helgolandicus* in the Bay of Biscay. Parasitism by *Ellobiopsis* sp. has the potential to reduce the fecundity of copepods: a reduction in size of both the seminal vesicle and the developing spermatophore sac was noted in parasitized males of *C. helgolandicus* (Albaina and Irigoien, 2006). The occurrence of epizoic filter-feeding Protozoa (*Vorticella* and *Zoothamnium*) and parasitic Protozoa (*Ellobiopsis*) on Calanoida was noticed in the Gulf of Gdańsk in 1998, 1999 and 2006. The relatively high (4–16% of all calanoids) level of infestation varied depending on the type of infestation (0.1–13% of the population of particular taxa). The dominant copepods – *Acartia* spp., *Temora longicornis* and *Centropages hamatus* – were attacked the most frequently (from 10.5% to 54% of all infested calanoids). Epibiosis and parasitism were observed on all copepod developmental stages (adults, juveniles and nauplii). Epibionts and parasites were located on different parts of the body, but mainly on the prosome. Infestation by epibionts and parasites was not restricted to calanoid copepods: it was also detected in non-negligible numbers on other crustaceans, namely, Harpacticoida, Cladocera (*Bosmina* sp.) and Cirripedia larvae (nauplii) in the Gulf of Gdańsk."

Outbreaks of cholera are sometimes correlated with copepods, which are common hosts of *Vibrio cholerae* (Colwell, 2004, Piasecki et al., 2004, Lee et al., 2007). Cholera outbreaks tend to be associated with algal blooms and the rapid increase in copepods that follows (Piasecki et al., 2004). *Eurytemora* spp. are known to host *V. cholerae* and are the most common of known copepod hosts in the Chesapeake Bay, where this has been studied (Colwell, 2004).

Eurytemora affinis has the ability to consume cyanobacteria and other toxic algal blooms; studies in the Baltic Sea indicate that this is likely an important mechanism of the biomagnification of toxins in organisms of economic importance, such as shrimp and fish (Engström et al., 2000, Karjalainen et al., 2008, Setälä et al., 2009).

The migratory and blood-feeding activities of the nematode *Anguillicola crassus* can cause extensive damage to the swim bladder of European eels, resulting in alterations in gas composition and reduction of swimming performance. Transmission in estuaries is ecologically possible since the copepod *Eurytemora affinis* is susceptible to infection acting as an intermediate host (Kirk et al., 2000; and references therein). The transmission cycle is not confined to fresh water and can be completed in estuarine (50% seawater) and marine (100% seawater) conditions. There are several eel species in the assessment region, but of unknown importance for the region (Stewart and Lockhart, 2005).

Q14- Genetic impact (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora affinis* would have low or no genetic impact on other species in the RA area. There are co-occurring native copepods of the same genus in the RA area. It is highly unlikely that it can interbreed with other species given that sometimes it cannot even reproduce with other clades of the same species in the field.

Background Information: *Eurytemora affinis* is composed of clades that are morphologically indistinguishable (Lee and Frost, 2002), yet with large genetic distances and idiosyncratic patterns of reproductive isolation among the clades (Lee, 2000). Laboratory cross-hybridization studies between two divergent clades of *E. affinis* from Europe and North America indicated that hybrids in the second generation were sterile (Sukhikh and Alekseev, 2013; and references therein).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora affinis* would have high impacts in few areas or moderate impacts in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be moderately impacted given the moderate effects of this copepod on ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. Given that bowhead whales (*Balaena mysticetus*) are plankton feeders, any impacts on plankton communities may impact this species. The probability or magnitude of such cascading effects are unknown. Likewise, potential impacts on larval species of at risk fish are also undetermined. As its invasion could bring cascading effects at higher trophic levels, species of fish at risk including: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and the thorny skate, *Amblyraja radiata* could be affected.

Q16- Impact on fisheries (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora affinis* would have high impact in few areas or moderate impact in many areas on fished species in the RA area. There is information on the ecological role of the species as an omnivorous predator/grazer from which impacts of this species to fish populations, communities and ecosystems in the RA area can be extrapolated. Much of this is extrapolated from freshwater systems.

Background Information: *Eurytemora affinis* could be a significant prey item for fish and other planktivores. Thorp and Casper (2003) demonstrated this potential in an enclosure experiment with yellow perch (*Perca flavescens*) in the St. Lawrence River; 99% of *E. affinis* disappeared from fish enclosures, presumably due to predation. *E. affinis* is an important component of the estuarine food web, including its role as a food source for higher trophic levels (Strasser et al., 2011; and references therein). Because *E. affinis* has become an abundant grazer in parts of the Great Lakes, it is possible that it has had important impacts on the food web—both adverse and beneficial (Lee et al., 2007).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eurytemora affinis* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: *Eurytemora affinis* (Poppe, 1880), native to the Ponto-Caspian region, is another example of euryhaline copepod species that has been reported from the western European coast, parts of Asia, and within North America from the Atlantic coast including the Gulf of Mexico to the Pacific coast (Kipp and Benson, 2010). This species represents a set of cryptic species in the northern hemisphere (Lee, 1999) and was recently defined as a sibling species among copepods in the Baltic Sea (Alekseev et al., 2009). One of the cryptic species inhabiting North America was recently described as the new species *Eurytemora carolleeae* Alekseev and Souissi, 2011. *Eurytemora affinis* has invaded many freshwater habitats in North America, Europe, and Asia (Lee, 1999). *Eurytemora affinis* originating from saline populations in the St. Lawrence River drainage is now established in all of the North American Great Lakes, where it is a dominating species in bays and harbor (Mills et al., 1993, Lee, 1999, 2000). All previous records of the species in North America should now be checked again to avoid mixing the newly described *Eurytemora carolleeae* with *E. affinis* that recently invaded from Europe (Alekseev and Souissi, 2011; and references therein). There have been at least eight independent invasions from saline into freshwater habitats on three continents, in North America, Europe and in Asia (Lee, 1999). These invasions into freshwater bodies likely occurred predominantly through ballast or bilge water transport, within the waterways connecting the saline and freshwater habitats, and in some cases through the transport of fish, such as striped bass, from downstream estuaries (Anderson and Clayton, 1959, Lee, 1999). Introductions of *E. affinis* into the Great Lakes originated from local estuaries (likely via ship ballast water), rather than from the brackish waters of the Black and Caspian Seas (Lee, 2016).

Eurytemora carolleeae Alekseev & Souissi, 2011

Phylum: Arthropoda
Class: Hexanauplia
Order: Calanoida
Family: Temoridae

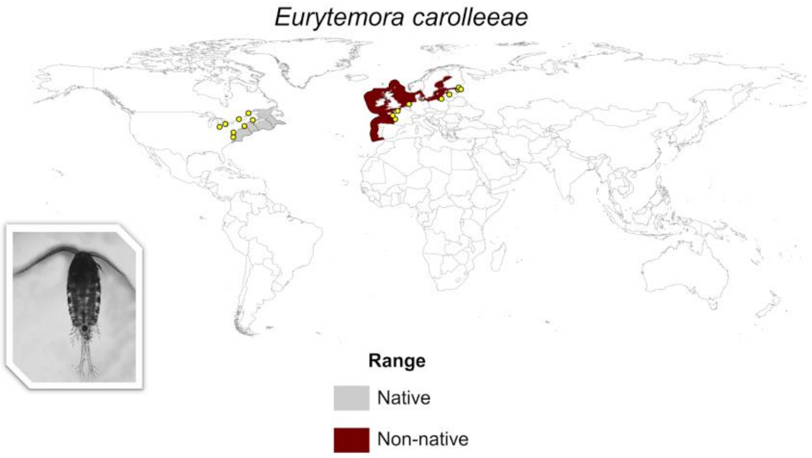

Figure 30: Ecoregions where Eurytemora carolleeae is distributed: native (grey) and non-native (dark red) regions. These regions are only representative to their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Eurytemora carolleeae occurrence points were obtained from GBIF.org (https://doi.org/10.15468/dl.m8a8wd, 17 December 2020), Sukhikh and Alekseev (2013), Alekseev and Souissi (2011). Note: this species can also be present in freshwater ecosystems, so inland points are included on the map. Picture of E. carolleeae modified from https://invasions.si.edu/nemesis/species_summary/85863.

CMIST scores for *E. carolleeae:*

Mean adjusted Likelihood of Invasion: 2.33

Mean adjusted Impact of Invasion: 1.98

Mean adjusted Overall CMIST score: 4.61

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: This is a new species that belongs to the Atlantic clade of the *Eurytemora affinis* complex outlined by previously published molecular work (Alekseev and Souissi, 2011). It has not previously been found in the Hudson Complex (Dispas, 2019). The new species was also found in Canada (St. Lawrence Estuary) and as an invasive species in the Baltic Sea (Alekseev and Souissi, 2011). It is possibly widely distributed along the North American Atlantic coast, as well as in inland waters from the Great Lakes to Mexico (Alekseev and Souissi, 2011).

Q2- Rate of introduction (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* could arrive frequently in high numbers in the RA area. The species has been found on arriving vessels and is present in a number of connected port ecoregions.

Background Information: *Eurytemora carolleeae* is present in the St-Lawrence estuary, and it has been identified in the ballast water tanks in the MV *Arctic*, which regularly transits from Quebec City to Raglan Mines in Deception Bay (Tremblay, 2017). Deception Bay normally receives the highest average annual domestic arrivals among Canadian vessels that conducted ballast management activities from 2005 to 2014 (Goldsmit et al., 2019). Tremblay (2017) and collaborators observed that *E. carolleeae* was in high abundance in the MV *Arctic*'s tanks in the Quebec City harbour (maximum in August at 15,774 ind. m^3) and it was still present in the control tanks of each trip upon arrival in Deception Bay (maximum in August at 1,585 ind. m^3).

Q3- Survival (suitable habitat) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests most of the RA area offers suitable habitat for *Eurytemora carolleeae.* It can live in a wide range of habitats which occur through much of the RA area. Reliable information is available for a closely related species (*E. affinis*).

Background Information: The species complex *Eurytemora affinis* has a geographic distribution from subtropical to subarctic regions that spans North America, Asia, and Europe, with habitat types that range from hypersaline salt marshes and brackish estuaries to completely fresh water (Saunders, 1993, Lee, 1999). *Eurytemora carolleeae* is a euryhaline species, and shows a tendency to advance further inland in continental waters (Lee and Petersen, 2003, Suarez-Morales et al., 2008).

From NEMESIS: "*Eurytemora carolleeae* is a planktonic calanoid copepod, recently separated from the circumboreal *E. affinis* species complex. Its native range extends from the St. Lawrence River, Canada to the St. Johns River, Florida. It has been introduced to the Pacific coast of North America in San Francisco Bay, California and Grays Harbor, Washington. It is also known from the Great Lakes and the Gulf of Finland in the Baltic Sea. It is found in salt marshes and estuaries with low-salinity waters. It is abundant in low-salinity waters in San Francisco Bay, where it is a grazer on phytoplankton populations and a major food source for larval fish." (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=85863)

Q4- Survival (suitable environment) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests most of the RA area offers suitable environmental conditions for *Eurytemora carolleeae.* It can survive in locations at sub-zero temperatures and it has been shown to survive well at 0° C. Even though there are reliable environmental data layers for the RA area to show a match, there is uncertainty in taxonomy and may have high levels of variability in physiological tolerance/species requirements among populations, similar to the closely related *E. affinis*.

Background Information: This species is able to withstand salinities ranging from 0 to 40 PSU and temperatures ranging from 0 to 30°C (Bradley, 1978, Lee and Petersen, 2003, Lee et al., 2013). It is able to survive and reproduce in freshwater, brackish, as well as marine habitats

(Lee, 2000). It is present in the St-Lawrence estuary (Alekseev and Souissi, 2011), which experiences extended periods with sub-zero temperatures (-1.5°C) similar to what would be expected under ice in the assessment region for much of the winter.

Q5- Establishment (reproductive requirements) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest reproductive requirements are sometimes available for *Eurytemora carolleeae* in the RA area. Species appear to tolerate a wide range of conditions and limited experimental studies show that reproductive performance was maintained under acute salinity increase and temperature decrease, albeit over a relatively small shift in conditions.

Background Information: The species is able to survive and reproduce in freshwater, brackish, and marine habitats (Lee, 2000). Cabrol et al. (2020) described *E. carolleeae* as a generalist species after having observed no significant decrease in reproductive performance under sudden changes in environmental conditions. There is a chance that diapause eggs could survive arctic winter conditions and hatch in the summertime, under milder conditions. Summer temperatures in the Hudson Strait close to Deception Bay vary in the water column between -1 and 5°C (Estrada et al., 2012). Hudson Strait annual subsurface (40-60 m) temperatures range between 4 and -1,8°C, with negative temperatures (< 0 ≥ -1,8°C) occurring from November until August (Straneo and Saucier, 2008). Thus, there might be a chance that *E. carolleeae* could survive and reproduce if discharged in sufficient amounts in the port of destination (Tremblay, 2017).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to-suggest natural control agents could slow population growth in the RA area for *Eurytemora carolleeae.* There are species in the RA area (e.g., mysids and herring) that could act as predators, but there is no information to indicate they could severely restrict population growth. There is only information on predators for a closely related species (*E. affinis*).

Background Information: *Eurytemora affinis* is a preferred prey for dominant zooplanktivores such as herring, sprat, mysids and predatory cladocerans (Lehtiniemi and Gorokhova, 2008; and references therein). Decreased food availability that can be caused by competition from other invasives such as the clam *Corbula amurensis*, can narrow the realized salinity niche of *E. affinis* (Hammock et al., 2016).

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* has the capacity for a wide range of natural dispersal in the RA area. There is good evidence for widespread potential given diapausing eggs and planktonic larvae/adults, however information is based on the closely related species *E. affinis*.

Background Information: For the closely related *E. affinis*: Under stressful conditions, diapausing eggs of *E. affinis* can survive in the sediment at least for several decades (Katajisto, 1996) and could be injected into the environment at a later time through disturbance of sediments via storm events and upwelling (Lee and Bell, 1999; and references therein).

Although the species (*E. affinis*) is a weak swimmer (Lee, 2016), in the HB LME, alongshore currents (Granskog et al., 2011) could disperse various life stages of species (eggs, larvae, adults) counterclockwise around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is solid evidence of widespread dispersal through ballast and the species has been observed in ballast of ships arriving in the RA area, where it can be released. There is some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: *Eurytemora carolleeae* is present in the St. Lawrence estuary, and it has been identified in the ballast water tanks in the MV *Arctic*, which regularly transits from Quebec City to Raglan Mines in Deception Bay (Tremblay, 2017). Deception Bay normally receives the highest average annual domestic arrivals among Canadian vessels that conducted ballast management activities from 2005 to 2014 (Goldsmit et al., 2019). Ballast transported by domestic coastal resupply and other boat traffic, could spread *E. carolleeae* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Recent invasions by *E. affinis* from saline estuaries and salt marshes into inland freshwater lakes and reservoirs have been mediated by human activity (Saunders, 1993, Lee, 1999, Winkler et al., 2008). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* would have high impacts in few areas or moderate impacts in many areas on population growth in the RA area. There is some evidence of impacts to protozoan ciliates and rotifer populations and the food web/ecosystems through grazing in the closely related *E. affinis.* The species also has a similar ecology to the more extensively studied *Acartia tonsa* which has been shown to have high impacts. Certainty is moderate because most of the information is from freshwater systems or inferred from other plankton species with similar ecologies.

Background Information: The impact of this species on ecosystems is not yet understood (Fofonoff et al., 2003, Tremblay, 2017) but inferred from closely related species and freshwater systems.

Q10- Impact on communities (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* would have high impacts in few areas or moderate impact in many areas on communities in the RA area. There is some evidence of impacts to protozoan ciliates and rotifer populations/communities and the food web/ecosystem through grazing in closely related *E. affinis* and it has a similar ecology to the more extensively studied *Acartia tonsa* which has been shown to have high impacts. Most of the information is from freshwater systems or inferred from other plankton species with similar ecologies.

Background Information: Considering the key role of *E. affinis* in the Baltic Sea zooplankton community, the invasion of *E. carolleeae* might have consequences for biodiversity, biogeography, conservation and ecosystem management in this area (Sukhikh and Alekseev, 2013)). Although, impacts on recipient ecosystems is not yet understood (Fofonoff et al., 2003, Tremblay, 2017).

Q11- Impact on habitats (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora carolleeae* would have high impacts in few areas or moderate impacts in many areas on habitat in the RA area. Information is lacking on habitat impacts, though it does not have physical impacts on the water column per se - only on the organisms within it. There is no evidence of increasing water clarity, but the possibility exists.

Background Information: Impacts on recipient ecosystems is not yet understood (Fofonoff et al., 2003, Tremblay, 2017).

Q12- Impact on ecosystem function (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* would have high impacts in few areas or moderate impacts in many areas on ecosystem function in the RA area. There is some evidence of impacts to protozoan ciliates and rotifer populations/communities and the food web/ecosystem through grazing in closely related *E. affinis* and it has a similar ecology to the more extensively studied *Acartia tonsa* which has been shown to have high impacts. Most of the information is from freshwater systems or inferred from other plankton species with similar ecologies.

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Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora carolleeae* would have high impact in few areas or moderate impact in many areas associated with diseases, parasites or travelers in the RA area. Much of the general information is on epibionts and parasites of copepods, which indicates that a relatively broad suite of taxa can be transported on a wide range of copepod taxa, including ones found in the RA area.

Background Information: *Eurytemora affinis* is a major host of many pathogens, including *Vibrio cholerae*, *Vibrio vulnificus*, and *Vibrio parahaemolyticus* (Huq et al., 1983, Heidelberg et al., 2002, Colwell, 2004). Huq et al. (2005) demonstrated a significant correlation between zooplankton blooms and cholera cases in Bangladesh. *Vibrio cholerae* occurs in the gut and on the surface of zooplankters such as *E. affinis* (Rawlings et al., 2007). Hatching nauplii can affect *V. cholerae* attachment to both adults and eggs. The presence of hatched nauplii resulted in eggs with greater numbers of *V. cholerae* (Rawlings et al., 2007). *Eurytemora affinis* is inhabited by an enormous consortium of microbiota. Interestingly, there is a dramatic shift in the copepod microbiome during invasions, with parallel shifts in the microbiome composition during saline to freshwater invasions (Gelembiuk, 2015). In lab conditions, *E. affinis* fed actively on the toxic *Nodularia spumigena* even when other food items were present, with no significant effect on their survival, feeding and reproduction (Kozlowsky-Suzuki et al., 2003). *Eurytemora affinis* would be capable of degrading/detoxifying toxins, but with some delay after ingestion (Kozlowsky-Suzuki et al., 2003). This process could vary depending on the conditions (e.g. starvation or transfer to a non-toxic food source) to which the animals are subjected after exposure to a toxic diet (Svensson, 2000). Only a very small fraction of the calculated ingested toxin could be found in the animals. Thus, even though these grazers might act as a link transferring toxins to higher trophic levels, the relative importance of this indirect pathway seems limited (Kozlowsky-Suzuki et al., 2003).

Bielecka and Boehnke (2014) did a study on epibionts and parasites of several copepods and stated that : "Epibiosis and parasitism are widespread in the zooplankton communities of marine and brackish environments (Hirche, 1974, Ho and Perkins, 1985, Timofeev, 1997, Hu and Song, 2001, Visse, 2007). Epibiotic overgrowth and parasitic infestation most often affect pelagic Copepoda (Wiktor and Krajewska-Sołtys, 1994, Timofeev, 2002, Visse, 2007, Walkusz and Rolbiecki, 2007). Both parasitism and epibiosis are considered harmful to planktonic animals.

Overgrowths of epizoic Protozoa can reduce swimming speed in Copepoda, especially when the antennae are heavily infested. Heavily-infested specimens are also more visible to predators, becoming easy prey for planktivorous animals (Chiavelli et al., 1993, Visse, 2007). Kimmerer and McKinnon (1990) described cases of *Paracalanus indicus* infested with parasitic Dinoflagellata (*Atelodinium* sp.) in the Indian Ocean. They reported that dinoflagellates formed a plasmodium that wrapped around the host's body, leading to its death. Other authors examined the effect of the parasite *Ellobiopsis* on the fecundity of *Calanus helgolandicus* in the Bay of Biscay. Parasitism by *Ellobiopsis* spp. has the potential to reduce the fecundity of copepods: a reduction in size of both the seminal vesicle and the developing spermatophore sac was noted in parasitized males of *C. helgolandicus* (Albaina and Irigoien, 2006). The occurrence of epizoic filter-feeding Protozoa (*Vorticella* and *Zoothamnium*) and parasitic Protozoa (*Ellobiopsis*) on Calanoida was noticed in the Gulf of Gdańsk in 1998, 1999 and 2006. The relatively high (4–16% of all calanoids) level of infestation varied depending on the type of infestation (0.1–13% of the population of particular taxa). The dominant copepods – *Acartia* spp., *Temora longicornis* and *Centropages hamatus* – were attacked the most frequently (from 10.5% to 54% of all infested calanoids). Epibiosis and parasitism were observed on all copepod developmental stages (adults, juveniles and nauplii). Epibionts and parasites were located on different parts of the body, but mainly on the prosome. Infestation by epibionts and parasites was not restricted to calanoid copepods: it was also detected in non-negligible numbers on other crustaceans, namely, Harpacticoida, Cladocera (*Bosmina* sp.) and Cirripedia larvae (nauplii) in the Gulf of Gdańsk."

The migratory and blood-feeding activities of the nematode *Anguillicola crassus* can cause extensive damage to the swim bladder of European eels, resulting in alterations in gas composition and reduction of swimming performance. Transmission in estuaries is ecologically possible since the copepod *Eurytemora affinis* is susceptible to infection acting as an intermediate host (Kirk et al., 2000; and references therein). The transmission cycle is not confined to fresh water and can be completed in estuarine (50% seawater) and marine (100% seawater) conditions. There are several eel species in the assessment area, but they are of unknown importance for the region (Stewart and Lockhart, 2005).

Q14- Genetic impact (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Eurytemora carolleeae* would have low or no genetic impact on other species in the RA area. There are co-occurring native copepods of the same genus in the RA area. It is highly unlikely that it can interbreed with other species given that sometimes the closely related *E. affinis* cannot even reproduce with other clades of the same species in the field.

Background Information: Laboratory cross-hybridization studies between two divergent clades of *E. affinis* from Europe and North America indicated that hybrids in the second generation were sterile (Sukhikh and Alekseev, 2013; and references therein). *Eurytemora affinis* is composed of clades that are morphologically indistinguishable (Lee and Frost, 2002), yet with large genetic distances and idiosyncratic patterns of reproductive isolation among the clades (Lee, 2000).

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora carolleeae* would have high impacts in few areas or moderate impacts in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be moderately impacted given the moderate effects of this copepod on ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay, and at-risk fish and mammal species are unlikely to be affected. Given that bowhead whales (*Balaena mysticetus*) are plankton feeders, any impacts on plankton communities may impact this species. The probability or magnitude of cascading effects is unknown. Likewise, the potential impacts on larval species of at risk fish are unknown. As its invasion could bring cascading effects at higher trophic levels, species of fish at risk including: northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and the thorny skate, *Amblyraja radiata* could be affected.

Q16- Impact on fisheries (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Eurytemora carolleeae* would have high impacts in few areas or moderate impacts in many areas on fished species in the RA area. There is information on the ecological role of the closely related *E. affinis* as an omnivorous predator/grazer from which impacts of this species to fish populations, communities and ecosystems in the RA area can be extrapolated. Much of this is extrapolated from freshwater systems.

Background Information: The closely related *Eurytemora affinis* can be a significant prey item for fish and other planktivores. Thorp and Casper (2003) demonstrated this potential in an enclosure experiment with yellow perch (*Perca flavescens*) in the St. Lawrence River; 99% of *E. affinis* disappeared from fish enclosures, presumably due to predation. *Eurytemora affinis* is an important component of the estuarine food web, including its role as a food source for higher trophic levels (Strasser et al., 2011; and references therein). Because *E. affinis* has become an abundant grazer in parts of the Great Lakes, it is possible that it has had important impacts on the food web—both adverse and beneficial (Lee et al., 2007).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Eurytemora carolleeae* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: This is a new species that belongs to the Atlantic clade of the *E. affinis* complex as outlined by previously published molecular work (Alekseev and Souissi, 2011). The new species was also found in Canada (St. Lawrence Estuary) and as an invasive species in the Baltic Sea (Alekseev and Souissi, 2011). The frequency of *E. carolleeae* in the Gulf of Riga was small, which could be the result of a relatively recent invasion and initial stages of invasive *Eurytemora* naturalization. In that case, the establishment of *E. carolleeae* in this area could be possible in the near future (Sukhikh and Alekseev, 2013). Invasion of *E. carolleeae* into the Gulf of Finland most likely arose from a subset of populations from the Atlantic clade. The Atlantic coastline of the USA is probably the origin of this North American species (i.e. *E. carolleeae*) (Sukhikh and Alekseev, 2013). Its invasion of the Great Lakes was explained by the opening of the St. Lawrence Seaway, followed by its introduction via ballast water (Pinel-Alloul et al., 2011, Vasquez et al., 2016).

ZOOPLANKTON – MACROZOOPLANKTON

Aurelia limbata Brandt, 1835

Phylum: Cnidaria
  Class: Scyphozoa
    Order: Semaeostomeae

Family: Ulmaridae

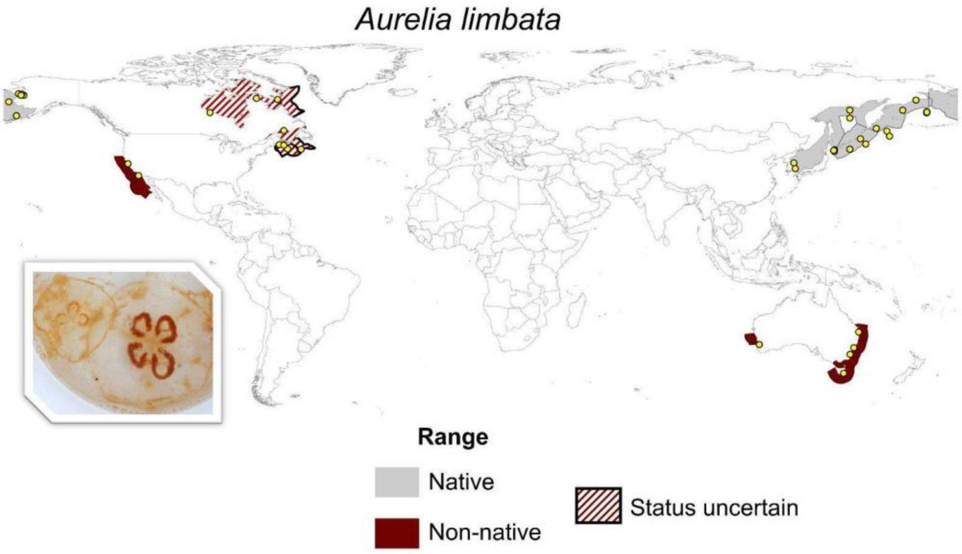

Figure 31: Ecoregions where Aurelia limbata is distributed. Consider that A. limbata is a species complex and that it can bring some confusion in identifying its status where it is distributed. For simplicity reasons, most likely native range (grey) has been identified while suspected non-native range (plain red and red stripes) should be carefully considered (see Q1, Q14 and Q17). These regions are only representative of its main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Note that points in the Eastern Canadian Arctic were mainly detected through metabarcoding (Chain et al., 2016) and found in recent surveys in Churchill (Howland et al. unpublished; Dispas, 2019), yet status is uncertain (red stripes). Aurelia limbata occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.s42l7g, 2 May 2017), NEMESIS (https://invasions.si.edu/nemesis/species_summary/-265), Miyake et al. (2002), Ki et al. (2008), Radchenko (2013), Chang et al. (2016), Zavolokin et al. (2008), Zavolokin (2010, 2011). Picture of A. limbata was provided by K. Howland.

CMIST scores for *A. limbata:*

Mean adjusted Likelihood of Invasion: 2.46

Mean adjusted Impact of Invasion: 2.30

Mean adjusted Overall CMIST score: 5.67

Q1- Present status in the area (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Aurelia limbata* is established in the RA area. Reads have been found in eDNA samples in ports from the Canadian Arctic.
This species has been found in successive recent surveys (since 2007) and shown to be present in substantially greater proportion of sampled sites over time between two sampling periods in 2007 and later in 2015 (Howland et al. unpublished). Also specimens collected in 2015, confirmed to be *A. limbata* based on barcoding of CO1 (Dispas, 2019).

Background Information: There are no historical reports in Hudson Bay for *A. limbata* (Stewart and Lockhart, 2005), but *Aurelia aurita* has been reported in southeastern Hudson Bay-James

Bay (Stewart and Lockhart, 2005). *Aurelia labiata* and *A. limbata* have been identified by metabarcoding (1 and 9560 reads, respectively) to be present in Churchill samples (Chain et al., 2016). *Aurelia limbata* is also present in Deception Bay (2 reads) and Iqaluit (10 reads) (Chain et al., 2016). *Aurelia aurita* has been found in eDNA samples taken in Churchill, however earlier studies only found *A. limbata*. The two are closely related and considered morphospecies. Either way, neither have previously been reported in Churchill aside from recent surveys (2007, 2011, 2015, Howland et al. unpublished; BOLD 2021, Dispas, 2019) despite being quite conspicuous. Several recent specimens collected in Churchill in 2015 were identified visually, photographed while fresh and later confirmed to be *A. limbata* based on barcoding using the COI marker (Howland & Winkler, unpublished). The species has not been reported elsewhere in the Canadian Arctic aside from 2 isolated records (one record in Lancaster Sound from 1928 by Kramp (1942) and one near Somerset Island from the Canadian Museum of Nature (1904) (https://www.gbif.org), however they are reported to be more widely distributed in the boreal waters of the Pacific region of Tohoku and Hokkaido, northern Japan, the Okhotsk Sea, Bering Sea, and the northern part of the Sea of Japan (Kishinouye, 1910, Uchida, 1954, Kramp, 1961, Larson and Harbison, 1990, Pogodin, 1998, Wrobel and Mills, 1998) where they are known to develop large blooms (Shibata et al., 2015, Goto, 2012) and have been infrequently reported off the southwest coast of Alaska. The common moon jellyfish is not a single species, *A. aurita*, but perhaps as many as 12 species (Dawson, 2003) with enough morphological similarity to confuse traditional taxonomists. The species is considered ‘cryptic’, since general morphological characters alone are not sufficient to differentiate between them (Graham and Bayha, 2008). Until recently, three generally accepted species had been recorded within the genus: the polar *A. limbata*, the north Pacific *A. labiata*, and the cosmopolitan *A. aurita* (Dawson and Jacobs, 2001). There are remarkably deep phylogenetic separations and the presence of at least 7 (and likely more) genetically isolated and ecologically distinguishable entities in *Aurelia*, confirming the earlier suggested demarcation of the *A. limbata* and *A. labiata* species from *Aurelia aurita* (Schroth et al., 2002).

Q2- Rate of introduction (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Aurelia limbata* could arrive frequently in low numbers or infrequently in high numbers in the RA area. It is widespread in connected ecoregions and there is at least one vector for transport.

Background Information: There is no information of the species in Chan et al. (2012), but looking at the species distribution, and knowing that the species could be transported either by ballast water and/or hull fouling and knowing the shipping traffic in the region, the species could arrive via shipping from ports in eastern North America. *Aurelia limbata* was found attached to two pieces of deep-sea debris, an aluminum beverage can and a plastic bottle, collected by bottom trawl at depths of 296 m and 392 m, respectively (Shibata et al., 2015) suggesting it could be moved as part of biofouling communities on vessels.

Q3- Survival (suitable habitat) (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests the RA area offers a moderate proportion of suitable habitat for *Aurelia limbata.* The species can live in places with ice cover. Reliable information is available but mostly for another closely related species (*Aurelia aurita*).

Background Information: Medusae of *A. limbata* occur in coastal shallow water areas and large numbers have been collected in depths up to 300 m (Miyake et al., 2011). Additionally, *A. limbata* medusae, which had planulae in brooding pouches on their oral arms, have been observed at depths of 200–400 m off Kushiro, Hokkaido, Northern Japan (Miyake et al., 2002). Miyake et al. (2002) speculated that this species can reproduce in the deep sea and that the polyps can inhabit such depths. The closely related *A. aurita* populations are found predominantly in coastal embayments, fjords, and estuaries where there are suitable substrata for the benthic scyphistoma polyp. Within these habitats, the degree of containment, tidal flow, water depth, temperature and salinity, and trophic condition can vary quite considerably (Lucas, 2001). Populations can occur in fjords that experience winter ice cover (Rasmussen, 1973, Hernroth and Gröndahl, 1983, 1985). Medusa abundance is generally higher in small, shallow, semi-enclosed or enclosed systems with limited tidal exchange than in open water systems or where depths exceed several hundreds of metres (Lucas, 2001; and references therein). Man-made hard substrates have the potential to increase the abundance of the *A. aurita* population (Janßen et al., 2013). *Aurelia aurita* is a highly flexible species that can adapt to a wide range of environmental conditions.

Q4- Survival (suitable environment) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable environmental conditions for *Aurelia limbata.* It is described as a cosmopolitan ecological generalist and it already occurs in the RA area. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: Strobilation of *Aurelia limbata* polyps was observed at 4°C without temperature change stimulation, raising the possibility that strobilation occurs in low-temperature environments throughout the year (Shibata et al., 2015). Strobilation of *A. limbata* has been observed with a temperature decrease from 15°C to 5–10°C (Straehler-Pohl and Jarms, 2010) and from 13–20°C to 4°C (Miyake et al. 2011). The water temperature and salinity remained within the range of 2.2–6.3°C and 33.4–33.9 psu, respectively, throughout the year, on the deep-sea floor where polyps were collected (Shibata et al., 2015). The closely related *A. aurita* can be considered both eurythermal and euryhaline in its distribution (Lucas, 2001). Increased jellyfish populations often are associated with warming caused by climate changes and possibly power plant thermal effluents. Jellyfish may benefit from eutrophication, which can increase small-zooplankton abundance, turbidity and hypoxia, all conditions that may favor jellyfish over fish (Purcell et al., 2007). Contrasting with these generalities, in the North Sea, Lynam et al. (2004) stated that *A. aurita* was found to be abundant after a cold event occurred between 1978 and 1982 that was related to decreased inflow of warm Atlantic water, increased inflow of cold, deep Norwegian water and arctic-boreal plankton. *Aurelia* spp. jellyfish, in particular, frequent highly eutrophic waters (Ishii, 2001, Mills, 2001, Nagai, 2003). *Aurelia aurita* has been described as a nearly cosmopolitan ecological generalist. However, new molecular and ecological data indicate *A. aurita* actually is a species-complex, comprising numerous locally adapted species (Dawson and Martin, 2001). *Aurelia aurita* s.l. lives in marine environments with widely different temperature and salinity ranges, including the Northwest Mediterranean Sea [12-26°C, salinity 37-39.2 (Fuentes et al., 2010)], the Red Sea [20.9-26.4°C, salinity 40.3-40.6 (Wolf-Vecht et al., 1992)], and the Baltic Sea [2-17°C, salinity 8-24 (Leppäranta and Myrberg, 2009)]. Each of these populations is genetically differentiated (Schroth et al., 2002) and may have different life cycle traits as an adaptation to their local habitat (Pascual et al., 2015).

Q5- Establishment (reproductive requirements) (Score = 3, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest reproductive requirements are almost always available for *Aurelia limbata* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >=4°C (temperature for asexual reproduction) resulted in selecting 2/3 of the RA area during summer months. Sexual production likely occurs between 2.2 and 6.3°C, suggesting that this species may reproduce sexually over the same area. The species requirements for asexual reproduction are known to some extent but the temperature requirements for sexual reproduction are conjecture based observations of mature adults with planulae at depths where it remains 2.2- 6.3°C.

Background Information: *Aurelia* spp. have complex life cycle, comprising an alteration of generations between an asexual benthic polyp, or scyphistoma, and a sexual pelagic medusae. Medusae of *A. limbata* occur in coastal shallow water areas and large numbers have been collected at depths up to 300 m (Miyake et al. 2011). Additionally, *A. limbata* medusae, which had planulae in brooding pouches on their oral arms, have been observed at depths of 200–400 m off Kushiro, Hokkaido, Northern Japan (Miyake et al., 2002). Miyake et al. (2002) speculated that this species can reproduce in the deep sea and that the polyps can inhabit such depths. Strobilation of *Aurelia limbata* polyps was observed at 4 °C without temperature change stimulation, raising the possibility that strobilation occurs in low-temperature environments throughout the year (Shibata et al., 2015). Strobilation of *A. limbata* has been observed with temperature decreases from 15°C to 5–10°C (Straehler-Pohl and Jarms, 2010) and from 13–20°C to 4°C (Miyake et al. 2011). The water temperature and salinity remained within the range of 2.2–6.3°C and 33.4–33.9, respectively, throughout the year, on the deep-sea floor where *A. limbata* polyps were collected (Shibata et al., 2015). Environmental temperature has apparently been a determining factor favoring ecological diversification in *Aurelia* spp. In cooler habitats *Aurelia* spp. may have adapted to lower temperatures for the onset of strobilation and to lower strobilation rates. The reduced efficiency of vegetative propagation likely is a consequence of the reduced, temperature dependent metabolism (Schroth et al., 2002). The ability of gelatinous species to occur in large numbers (i.e. to bloom) is due to the cnidarians having both asexual and sexual reproduction (Purcell et al., 2007). Polyps in the field can survive for prolonged periods of many months in the encysted state (Brewer and Feingold, 1991). The main period of strobilation usually occurs following a reduction in temperature (Lucas, 2001). Both temperature and salinity can have significant effects on asexual reproduction (Purcell et al., 2007). Increased light also accelerated strobilation in *A. labiata* (Loeb, 1973). When environmental conditions do not promote sexual reproduction, scyphistoma reproducing asexually (in more than half-a-dozen ways) may maintain a population for several years (Dawson and Martin, 2001).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest natural control agents could slow population growth in the RA area for *Aurelia limbata.* There are closely related species in the RA area that could act as predators. There is limited information on predators and the available information is from another species in the same genus (*A. aurita*).

Background Information: Predators of *A. aurita* include other pelagic cnidarians such as *Cyanea capillata*, *Aequorea victoria* and *Phacellophora camtschatica*, turtles and various fish species (Lucas, 2001; and references therein). These may be potentially important regulators of medusa abundance where predator and prey species co-exist, but it is difficult to assess accurately in the field because jelly-fish are not easily identified in gut contents. There are also several parasites in *A. aurita* medusae, with larval trematodes and cestodes and hyperiid amphipods having received the most attention (Lucas, 2001; and references therein). Takao et al. (2014) found that of 78 species tested only eight were found to be potential predators of *Aurelia aurita* polyps in Japanese waters - these may possess some protective or adaptive mechanisms against nematocysts, so that they can ingest them and included five gastropod species (*Calliostoma unicum*, *Pleurobranchaea japonica*, *Hermissenda crassicornis*, *Sakuraeolis enosimensis* and *Sakuraeolis sakuracea*) and three crustacean species (*Rhynchocinetes uritai*, *Latreutes anoplonyx* and *Hyastenus diacanthus*). In particular, *Calliostoma unicum*, *Pleurobranchaea japonica*, *Hermissenda crassicornis*, *Rhynchocinetes uritai* and *Hyastenus diacanthus* consumed more than 300 polyps per predator per day in the laboratory.

From Dong (2019): “The relationships between gelatinous medusae and fish are complex due to varying interactions at different stages of their life cycles. For example, the medusae may predate on pelagic fish eggs and larvae, but fish also predate on gelatinous species. Meanwhile, the medusae compete with several small planktivorous fish for the same zooplankton prey

(Purcell and Sturdevant, 2001, Richardson et al., 2009). Therefore, some studies have suggested that overfishing may help facilitate jellyfish outbreaks (Lynam et al., 2006, Purcell et al., 2007, Richardson et al., 2009)."

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* has the capacity for a wide range of natural dispersal in the RA area. Although the species is shown to be dispersed widely, genetic evidence suggests limited dispersal.

Background Information: Medusae of *Aurelia* spp., persist for weeks to months. Diffusion processes, ocean currents, and active swimming could potentially disperse this stage over thousands of kilometers (e.g., Johnson et al., 2005). However, Dawson et al. (2005) showed that despite the high dispersal potential of *Aurelia* spp., a molecular phylogeny of the genus exhibits substantial biogeographic regionalization, indicating that genetic isolation is more common than previously recognized (Graham and Bayha, 2008).

The ability of gelatinous species to occur in large numbers (i.e. to bloom) is due to the cnidarians having both asexual and sexual reproduction (Purcell et al., 2007). In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait.

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is solid evidence of widespread dispersal through ballast waters, though some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: Dawson et al. (2005) note that one cryptic species (*Aurelia* sp. 1) with a global distribution is likely related to historical shipping activity. They concluded, based on this species' limited ability to traverse the Pacific Ocean, that its global distribution was invasive and mediated (possibly multiple times) by shipping (Dawson, 2003, Dawson et al., 2005; from Graham and Bayha, 2008). Another of their species (*Aurelia* sp. 4) was also identified as invasive in Hawaii from an Indo-Pacific origin (Dawson et al., 2005; from Graham and Bayha, 2008). Many species may easily survive transport because the benthic stages enter a dormant stage (cyst or stolon) in response to stressful conditions in which they can survive extended periods (e.g. Arai, 1997, Boero et al., 2002). Planulae of *A. aurita* are lecithotrophic, spending between 12 h and 1 week in the water column prior to settling (Lucas, 2001), which can be transported by ballast water. Polyps are found on the undersides of virtually any natural hard substrata, and they also can colonize glass, ceramic or plastic settling plates (Lucas, 2001), a characteristic that makes it easy to be transported by hull fouling. Hull fouling and ballast transported by domestic coastal resupply and other boat traffic could spread *Aurelia* spp. from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half the of total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* would have high impacts in many areas on population growth in the RA area. Based on general principles for *Aurelia* spp., the jellyfish genus seems to have large impacts on planktonic populations.

Background Information: *Aurelia* competes with plankton feeding fish or preys on their juveniles (Korsun et al., 2012). There is an inverse relationship between abundance of *A. aurita* with total zooplankton and copepod numbers in the Baltic Sea (Behrends and Schneider, 1995). It is considered a keystone species in control of trophic structures (Olesen, 1995), although in certain regions (the central Baltic Sea) it was found that *A. aurita* did not regulate the zooplankton community nor competed or preyed on fish larvae (Barz and Hirche, 2005).

Purcell and Sturdevant (2001) suggest that *A. limbata* preys mostly on copepods in an Alaskan bay.

There seems to have been large variability in the grazing impact on zooplankton depending on the seasonal abundance of *A. aurita* (Riisgård et al., 2012).

From Dong (2019): "Blooms of the moon jellyfish *Aurelia* spp. can have negative impacts on human economic activities, that is, clogging cooling water intakes at coastal power plants, causing mortality of aquaculture species, and interfering with fisheries and tourism (Richardson et al., 2009, Dong et al., 2010, Purcell, 2012). The relationships between gelatinous medusae and fish are complex due to varying interactions at different stages of their life cycles. For example, the medusae may predate on pelagic fish eggs and larvae, but fish also predate on gelatinous species. Meanwhile, the medusae compete with several small planktivorous fish for the same zooplankton prey (Purcell and Sturdevant, 2001, Richardson et al., 2009). Therefore, some studies have suggested that overfishing may help facilitate jellyfish outbreaks (Lynam et al., 2006, Purcell et al., 2007, Richardson et al., 2009)."

Q10- Impact on communities (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* would have high impacts in many areas on communities in the RA area. Based on general principles for *Aurelia* spp., the jellyfish genus seems to have large impacts on planktonic communities.

Background Information: Studies showed an inverse relationship between abundance of *A. aurita* with total zooplankton and copepod numbers in the Baltic Sea. Fine-filter feeders and raptorial feeders were much more important in years when medusae occurred in low densities, whereas coarse-filter feeders dominated in the opposite situation (Behrends and Schneider, 1995). Although in certain regions (central Baltic Sea) it has been shown that *A. aurita* did not regulate the zooplankton community nor competed or preyed on fish larvae (Barz and Hirche, 2005). When gelatinous predatory pressure on herbivores is sufficiently intense, and nutrients and light are not limiting, top-down regulation of plankton communities may also occur, resulting in increased phytoplankton blooms and alteration of species composition (Lucas, 2001; and references therein).

There seems to have been large variability in the grazing impact on zooplankton depending on the seasonal abundance of *A. aurita* (Riisgård et al., 2012).

From Dong (2019): "Blooms of the moon jellyfish *Aurelia* spp. can have negative impacts on human economic activities, that is, clogging cooling water intakes at coastal power plants, causing mortality of aquaculture species, and interfering with fisheries and tourism (Richardson et al., 2009, Dong et al., 2010, Purcell, 2012). The relationships between gelatinous medusae and fish are complex due to varying interactions at different stages of their life cycles. For example, the medusae may predate on pelagic fish eggs and larvae, but fish also predate on gelatinous species. Meanwhile, the medusae compete with several small planktivorous fish for the same zooplankton prey (Purcell and Sturdevant, 2001, Richardson et al., 2009). Therefore, some studies have suggested that overfishing may help facilitate jellyfish outbreaks (Lynam et al., 2006, Purcell et al., 2007, Richardson et al., 2009)."

Q11- Impact on habitats (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* would have high impacts in few areas or moderate impacts in many areas on habitat in the RA area. Based on general principles for *Aurelia* spp., the jellyfish genus seems to have large impacts on planktonic populations/communities/ecosystems, thus impacting water clarity and nutrients.

Background Information: Fine-filter feeders and raptorial feeders were much more important in years when medusae occurred in low densities, whereas coarse-filter feeders dominated in the opposite situation (Behrends and Schneider, 1995). When gelatinous predatory pressure on herbivores is sufficiently intense, and nutrients and light are not limiting, top-down regulation of plankton communities may also occur, resulting in increased phytoplankton blooms (Lucas, 2001; and references therein). These actions could change the habitat.

There seems to have been large variability in the grazing impact on zooplankton depending on the seasonal abundance of *A. aurita* (Riisgård et al., 2012).

From Dong (2019): "Blooms of the moon jellyfish *Aurelia* spp. can have negative impacts on human economic activities, that is, clogging cooling water intakes at coastal power plants, causing mortality of aquaculture species, and interfering with fisheries and tourism (Richardson et al., 2009, Dong et al., 2010, Purcell, 2012). The relationships between gelatinous medusae and fish are complex due to varying interactions at different stages of their life cycles. For example, the medusae may predate on pelagic fish eggs and larvae, but fish also predate on gelatinous species. Meanwhile, the medusae compete with several small planktivorous fish for the same zooplankton prey (Purcell and Sturdevant, 2001, Richardson et al., 2009). Therefore, some studies have suggested that overfishing may help facilitate jellyfish outbreaks (Lynam et al., 2006, Purcell et al., 2007, Richardson et al., 2009)."

Q12- Impact on ecosystem function (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* would have high impacts in few areas or moderate impact in many areas on ecosystem function in the RA area. Based on general principles for *Aurelia* spp., the jellyfish genus seems to have large impacts on planktonic ecosystems, and is considered a keystone species, impacting water clarity, nutrients and trophic structure. However, this is conditional on intensity of jellyfish blooms and effects depend on sufficient predation on herbivores.

Background Information: When gelatinous predatory pressure on herbivores is sufficiently intense, and nutrients and light are not limiting, top-down regulation of plankton communities may also occur, resulting in increased phytoplankton blooms and alteration of species composition (Lucas, 2001; and references therein). *Aurelia* spp. are considered to be keystone species in the control of trophic structures (Olesen, 1995).

There seems to have been large variability in the grazing impact on zooplankton depending on the seasonal abundance of *A. aurita* (Riisgård et al., 2012).

From Dong (2019): "Blooms of the moon jellyfish *Aurelia* spp. can have negative impacts on human economic activities, that is, clogging cooling water intakes at coastal power plants, causing mortality of aquaculture species, and interfering with fisheries and tourism (Richardson et al., 2009, Dong et al., 2010, Purcell, 2012). The relationships between gelatinous medusae and fish are complex due to varying interactions at different stages of their life cycles. For example, the medusae may predate on pelagic fish eggs and larvae, but fish also predate on gelatinous species. Meanwhile, the medusae compete with several small planktivorous fish for the same zooplankton prey (Purcell and Sturdevant, 2001, Richardson et al., 2009). Therefore, some studies have suggested that overfishing may help facilitate jellyfish outbreaks (Lynam et al., 2006, Purcell et al., 2007, Richardson et al., 2009)."

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* would have high impacts in few areas or moderate impacts in many areas associated with diseases, parasites or travelers in the RA area. There are a wide variety of parasites that could be transported and infect fish in the RA area, which is well demonstrated for cnidarians more generally, including Scyphozoans. These could affect the fitness and general health of native fish and cnidarians in the RA area if introduced with *A. limbata*, but extent of impacts are uncertain.

Background Information: Trematode, cestode and nematode larvae are widely distributed in pelagic Hydrozoa, Scyphozoa and Ctenophora (Lauchner, 1980a, b). It is not known to what extent fish acquire the parasites by eating coelenterates rather than eating other possible intermediate hosts (Purcell and Arai, 2001).

From Kondo et al. (2016): "Jellyfish not only play an important role as predators in the marine ecosystem, but they also function as prey and hosts for a wide variety of organisms (Arai, 1988, Purcell and Arai, 2001, Arai, 2005, Ohtsuka et al., 2009, Ohtsuka et al., 2010). Interactions between jellyfish and fish have long been known and have been comprehensively reviewed by many authors (Mansueti, 1963, Thiel, 1970, 1978, Arai, 1988, Ates, 1988, Purcell and Arai, 2001, Karplus, 2014, Kondo et al., 2014). Jellyfish are utilized by fish for school formation, food collection, and prey (Masuda, 2008, Masuda et al., 2008). In addition, endoparasitic helminths are transmitted from intermediate host jellyfish to definitive host fish via predation (Marcogliese, 1995). Some digenean trematodes are known to use cnidarians as their second intermediate hosts (Gomez Del Prado Rosas et al., 2000, Martorelli, 2001, Morandini et al., 2005) and/or paratenic hosts (Stunkard, 1969, Lauchner, 1980a, Stunkard, 1980b, a). Medusivorous fish become infected by trematodes through predation of infected jellyfish and act as definitive hosts (Bray et al., 1993, Ohtsuka et al., 2010)."

Q14- Genetic impact (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Aurelia limbata* would have high genetic impacts in few areas or moderate genetic impact in many areas on other species in the RA area. Although there is a high probability of hybridization within *Aurelia*, there are no known native *Aurelia* species found within the RA area or more broadly within the Canadian Arctic.

Background Information: There is high morphological variability in the medusa stage and as many as 12 *Aurelia* species or subspecies were described earlier (Mayer, 1910) (Kramp 1961, from Schroth et al 2002), of which only two species were recognized by taxonomists (Russell, 1970). Based on morphological data, the bulk of original species are summarized as a single species, *A. aurita*, which is deemed to be an ecological generalist and occupy worldwide habitats of all but north polar oceans. The second species, *A. limbata*, is seen as the boreal-arctic living counterpart (Schroth et al., 2002). Hybridization events between *A. aurita* and *A. labiata* have been due to anthropogenic mediated secondary contacts, e.g. by shipping ballast water (Greenberg et al., 1996). The compound action of hybridization and temperature dependent adaptations have contributed to the formation of the high species richness in *Aurelia* (Schroth et al., 2002).

Each clade of *Aurelia* constitutes a distinct species (Dawson and Jacobs, 2001). Recognizing these clades as species also is supported by the coincidence of geographic or habitat separation with probable reproductive and physiological isolation (Dawson and Jacobs, 2001). The majority of data available at this time suggest strongly that descriptions of *Aurelia* as a bi- or tri-typic genus in which most populations belong to one almost ubiquitous generalist, *A. aurita* (Dawson and Martin, 2001).

Although there is a high probability of hybridization within *Aurelia*, there are no known native *Aurelia* species found within the RA area or more broadly within the Canadian Arctic.

Note:There are two reports from James Bay, one of *Aurelia aurita* from the La Grande estuary (Grainger and McSween, 1976) and the other, for *Aurelia aurita limbata* in the Wemindji area (Bussières et al., 2008), but they are likely misidentifications.

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Aurelia limbata* would have high impacts in few areas or moderate impacts in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be moderately impacted given the moderate effects of this jellyfish on ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. As its invasion could bring cascading effects at higher trophic levels, species of fish at risk including northern wolffish, *Anarhichas denticulatus*; spotted wolffish, *Anarhichas minor*; and thorny skate, *Amblyraja radiata* could be affected, and mammals at risk such as bowhead whale (*Balaena mysticetus*) could be affected since it feeds on small planktonic crustaceans which are a prey species for *Aurelia* spp. (Stewart and Lockhart, 2005).

Q16- Impact on fisheries (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Aurelia limbata* would have high impacts in many areas on fished species in the RA area. This important group of jellyfish (*Aurelia* spp.) has known direct and presumed indirect (via predation of food of fish species) effects on fisheries and aquaculture species. Most information is from related species (mostly *A. aurita*).

Background Information: In general, jellyfish interfere with tourism by stinging swimmers, fishing by clogging nets, aquaculture by killing fish in net-pens and power plants by clogging cooling-water intake screens. They also have indirect effects on fisheries by feeding on zooplankton and ichthyoplankton, and, therefore, are predators and potential competitors of fish (Purcell et al., 2007). Large catches of jellyfish can split fishing nets and ruin the quality of catches (e.g., in Japan since 1990 when *A. aurita* increased) (Purcell et al., 2007). *Aurelia aurita* has been described to clog seawater intake screens of power and desalination plants causing power reductions and shutdowns in Japan (Purcell et al., 2007). Clogging incidents can cause emergency situations at nuclear power plants and can result in significant power loss and economic damage to affected cities. *Aurelia* competes with commercial plankton feeding fish or preys on their juveniles, creating economic impacts as well (Korsun et al., 2012). None of these impacts apply to the assessment region.

From Baxter et al. (2011): "*Aurelia aurita* can cause severe gill problems in marine-farmed fish. With aquaculture predicted to expand worldwide and evidence suggesting that jellyfish populations are increasing in some areas, this threat to aquaculture is of rising concern as significant losses due to jellyfish could be expected to increase in the future. Damage to fish in aquaculture may therefore be direct, through stinging of the skin or gills (if small individuals or loose nematocysts are inhaled), or indirect, through de-oxygenation of the surrounding water [2]. The impacts of jellyfish blooms on finfish in aquaculture are not exclusive to salmon production, and are likely to occur in all areas where *Aurelia* spp. and other jellyfish are common. Our data have global relevance, as jellyfish have affected or may potentially affect highly productive aquaculture operations."

From Dong (2019): “The moon jellyfish has many consequences. It is mostly distributed in coastal waters, which are also potential nursery areas for economically important fish species. It can interfere with fisheries through various means, including predation on ichthyoplankton (Möller, 1984, Duffy et al., 1997, Pereira et al., 2014), potential competition between *Aurelia* spp. and zooplanktivorous fish and fish larvae for prey (Purcell and Sturdevant, 2001), and hindrance of fishing activities (Uye and Ueta, 2004). Both field studies and laboratory studies have shown that *Aurelia* spp. can consume fish larvae of economically important species ((Möller, 1984, Duffy et al., 1997, Pereira et al., 2014). For example, in Kiel Fjord, young *Aurelia aurita* consumed large quantities of the yolk-sac stages of herring larvae (Möller, 1984). The rate of instantaneous mortality in the yolk-sac stages of *Sciaenops ocellatus* preyed on by *A. aurita* was 0.06±0.09/h (Duffy et al., 1997). Experimental studies have also shown that *A. aurita* preferentially selected copepods and fish eggs as prey (Pereira et al., 2014). The diet of *Aurelia labiata* includes copepods, cladocerans, and bivalve veligers, which overlaps with the diets of zooplanktivorous fish (Purcell and Sturdevant, 2001).

Negative interactions between moon jellyfish *Aurelia* spp. and aquaculture species have been reported in Europe, Australia, North America, and Asia (Willcox et al., 2008, Baxter et al., 2011, Rodger et al., 2011, Purcell et al., 2013, Dong et al., 2017). The possible damage by *Aurelia* spp. to aquaculture species may occur directly through stinging the skin or gills or indirectly through deoxygenation of the surrounding waters (Båmstedt et al., 1998). The stings of *Aurelia* spp. on aquaculture species have often been underestimated due to their mild stings on humans. However, several studies have reported that *Aurelia* spp. stings have caused severe damage to aquaculture species. For example, *A. aurita* caused a mass mortality of fish and bivalves in 1950 (Yasuda, 1988). *Aurelia* spp. caused mortalities of farmed salmon in Tasmania during summers from 1998 to 2001 (Willcox et al., 2008). Ephyrae and small medusa of *A. aurita* are thought to cause mortality of marine-farmed salmonids in European waters (Baxter et al., 2011, Purcell et al., 2013). Recently, blooms of *Aurelia* sp.1 ephyrae in the sea cucumber culture ponds of China have come to be regarded as the cause of sea cucumber vomiting (Dong et al., 2017)."

Q17- Past invasion history **(Score = 3, Certainty = 3)**

Score Rationale: Considerable reliable information suggests *Aurelia limbata* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Many authors considered all Moon Jellyfish to be a single species, *Aurelia aurita*. However, molecular studies now indicate that *'A. aurita'* is a complex of at least 13 species. *Aurelia* sp. 1 is a genetically and morphologically distinct form, with a Northwest Pacific origin. Molecular studies indicate that this species is most likely native from southern Korea (and possibly China) to northern Japan. Introduced populations have been reported from California, Australia, France, and Wales. Specific impacts of *Aurelia* sp. 1 are little known, because of their confusion with native cryptic species (https://invasions.si.edu/nemesis/browseDB/SpeciesSummary.jsp?TSN=-265). Based on this species' limited ability to traverse the Pacific Ocean, it is thought that its global distribution was invasive and mediated (possibly multiple times) by shipping (Dawson, 2003, Dawson et al., 2005). Another of their species (*Aurelia* sp. 4) was also identified as invasive in Hawaii from an Indo-Pacific origin (Dawson et al., 2005). A bloom in Tokyo by *A. aurita* was possibly caused by climate change, eutrophication, fishing, aquaculture, construction and invasion (Purcell et al., 2007; and references therein).

Mnemiopsis leidyi A. Agassiz, 1865

Phylum: Ctenophora

Class: Tentaculata

Order: Lobata

Family: Bolinopsidae

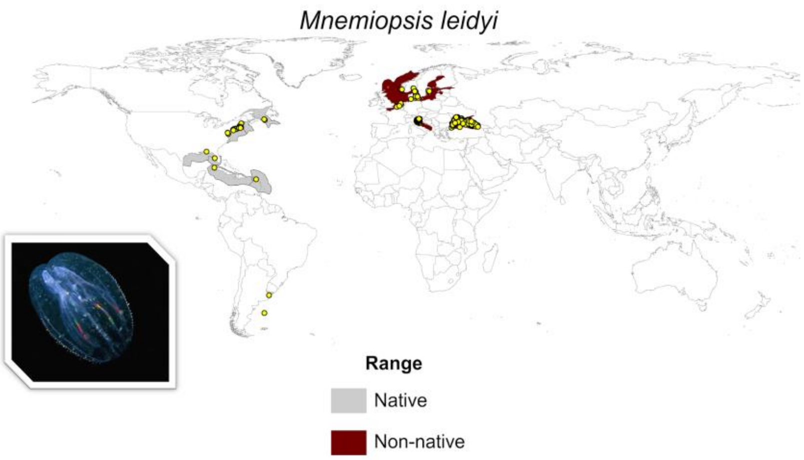

Figure 32: Ecoregions where Mnemiopsis leidyi is distributed: native (grey) and non-native (dark red) regions. These regions are only representative of their main known distribution and occurrence points (in yellow) have not been collected in an exhaustive manner. Mnemiopsis leidyi occurrence points were obtained from OBIS (https://obis.org/), GBIF.org (http://doi.org/10.15468/dl.ak6zel, 10 April 2017), Malej et al. (2017). Picture of M. leidyi modified from https://ocean.si.edu/ocean-life/invertebrates/sea-walnut-mnemiopsis-leidyi.

CMIST scores for *M. leidyi:*

Mean adjusted Likelihood of Invasion: 2.23

Mean adjusted Impact of Invasion: 2.54

Mean adjusted Overall CMIST score: 5.66

Q1- Present status in the area (Score = 1, Certainty = 2)

Score Rationale: Some reliable information suggests *Mnemiopsis leidyi* is not established in the RA area. However, survey effort is low relative to the size of the RA area.

Background Information: No reports were found from Hudson Bay (Stewart and Lockhart, 2005, Brown et al., 2016, Chain et al., 2016, Dispas, 2019).

Q2- Rate of introduction (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Mnemiopsis leidyi* could arrive frequently in low numbers or infrequently in high numbers in the RA area. The species is present in connected pathways, but not established in close geographic proximity, so natural spread is unlikely.

Background Information: There is no information on the species in Chan et al. (2012). There is potential transportation of this species via shipping from European ports into Canadian Arctic ports (looking at the species distribution and knowing that the species could be transported by ballast water (Graham and Bayha, 2008) and knowing the shipping traffic in the region).

Q3- Survival (suitable habitat) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests most of the RA area offers suitable habitat for *Mnemiopsis leidyi.* Coastal waters and estuaries are in most of the assessment area.

Background Information: The maximum depth where the species has been observed is 80-110 m, but maximum densities have been found between 40-60 m (Huwer et al., 2008). It is found mostly in coastal waters and estuaries and coincides with modelling results predicting survival in the North Sea (Collingridge et al., 2014).

Q4- Survival (suitable environment) **(Score = 2, Certainty = 3)**

Score Rationale: Considerable reliable information suggests the RA area offers a moderate proportion of suitable environmental conditions for *Mnemiopsis leidyi.* Although survival temperature is 2°C, it can withstand and occur in waters of -0.7°C, increasing the area where environmental variables are suitable. Calculated in ArcGIS with minimum sea surface temperature layer >= 2 shows a great area of Hudson Bay. Species requirements are well known and there are reliable environmental data layers for the RA area to show a match.

Background Information: From Collingridge et al. (2014): "Survival temperature 2ºC (Purcell et al., 2001), salinity 4.5 PSU (Sarpe et al., 2007), adult food 3 mg C/m^3 (Kremer, 1994, Purcell et al., 2001, Fuentes et al., 2010)." A polymorphic species with wide environmental tolerances and high phenotypic variability, it can live over a broad range of salinity and temperature conditions (https://www.cabi.org/isc/datasheet/75102). In the Barents Sea it has been found at temperatures below 10°C (Faasse and Bayha, 2006). *Mnemiopsis leidyi* may occur in temperatures from -0.7 to 35°C and in salinities between 3.4-70 ppt, but for shorter periods of time, slightly lower or higher values are tolerated (Miller, 1974, Hansson, 2006) (https://www.nobanis.org/globalassets/speciesinfo/m/mnemiopsis-leidyi/mnemiopsis_leidyi.pdf).

From DAISIE: "It is possible that *Mnemiopsis* has survived multiple winters in the southern North Sea estuaries, where winter temperatures average about 3°C (sometimes falling to ~2°C), with higher temperature refuge areas at depth (Wolff, 1973) Hydro Meteo Centrum Zeeland (2006) and it has survived winter conditions (although abundance decreased) in the Southern Baltic Sea (Kube et al., 2007). Modelling has shown that it could overwinter in the North Sea conditions (David et al., 2015). On the basis of observations Shiganova et al. (2001) noted an adverse effect of cold winter temperatures on the growth of the overwintering *Mnemiopsis* population in the Black Sea and subsequent limitation of their spring and summer populations. In contrast, Sullivan et al. (2001) and Costello et al. (2006) found no clear correlation between winter temperatures and the spring *Mnemiopsis* bloom date for its native range in Narragansett Bay USA, but suggested a significant correlation with spring temperatures. A similar link was also reported in Chesapeake Bay, USA (Purcell and Decker, 2005). There was a positive correlation between the Black Sea warmer spring surface mixed layer temperature and the *Mnemiopsis* biomass increase. The mixed layer temperature of 10°C therefore seems to be a threshold for high spring *Mnemiopsis* production in the Black Sea (Oguz et al., 2008). Three factors act in a hierarchy to determine the abundance of this comb jelly, with temperature being the most important ranging from 6°C in winter to 31°C in summer, food availability second, and mortality by predation third. It is invading waters of salinities ranging from 3‰ in the Sea of Azov to 39‰ in the eastern Mediterranean". (http://www.creaf.cat/delivering-alien-invasive-species-europe).

Q5- Establishment (reproductive requirements) **(Score = 2, Certainty = 2)**

Score Rationale: Some reliable information suggests reproductive requirements are sometimes available for *Mnemiopsis leidyi* in the RA area. To correctly evaluate the extension of suitable area, the following was calculated using ArcGIS: Maximum sea surface temperature >=12°C resulted in selecting a small region in James Bay. However, it has been seen that the species can reproduce at 4°C, which resulted in a moderate extension. Hence, it can be concluded that reproductive conditions sometimes are available in the RA area. There is some uncertainty regarding minimum temperature for reproduction and this may vary by population.

Background Information: From Collingridge et al. (2014): “The minimum temperature for reproduction is 12ºC (Sarpe et al., 2007, Lehtiniemi et al., 2012), and salinity 10 PSU (Jaspers et al., 2011, Lehtiniemi et al., 2012), although there are observations that the species reproduced at low salinities (6.5-9 PSU) and temperatures (4-7ºC) in the northern Baltic Sea (Lehtiniemi et al., 2007). It is a self-fertilizing hermaphrodite, preadapted to rapid colonization (Kremer, 1976). All the planktonic Ctenophora are simultaneous hermaphrodites capable of self-fertilization; therefore, a single adult individual may give birth to a viable posterity (Pianka et al., 1974, Reeve et al., 1978, Shiganova, 2000). The species has the ability to regenerate from fragments larger than one quarter of an individual (Coonfield, 1936). In its native areas and in the Black Sea and adjacent waters, population density seems not to be limited by salinity, but does decrease towards winter due to temperature decline (Purcell et al., 2001). In the northern Baltic, the most important factor reducing its abundance is probably decreasing food availability due to cooling of surface waters and decreasing plankton productivity towards late autumn and winter (Johansson et al., 2004,Viitasalo 1992, from Lehtiniemi et al 2007). It is assumed that low salinities in combination with low temperatures might cause higher mortalities, reduced growth rates and subsequently reduced population density increases in summer (Shiganova, 1998). Conditions in the summer in the North Sea are suitable for reproduction (Collingridge et al., 2014).

Q6- Establishment (natural control agents) (Score = 2, Certainty = 3)

Score Rationale: Considerable reliable information suggests natural control agents could slow population growth in the RA area for *Mnemiopsis leidyi*. There is the possibility of ballast transport for the parasite *Edwardsiella lineata*, which has considerable published information on its interactions.

Background Information: *Beroe ovata* is a ctenophore predator feeding on planktivorous comb jellies and *M. leidyi* above all. *Beroe ovata* is the best candidate to control *M. leidyi* population size as shown in the Black Sea. It controls its own population size by stopping reproduction in the absence of available prey; large adult individuals are eliminated and others stay near the bottom without movements until prey is available (Seravin et al., 2002). After the accidental introduction of *B. ovata* to the Black Sea, abundance of *M. leidyi* immediately dropped to levels so low that no further damage was inflicted. In fact, the ecosystem almost immediately began to recover (Molnar et al., 2008). *Beroe ovata* is not present in the assessment area (https://obis.org/). Another predator is the scyphomedusan *Chrysaora quinquecirrha* (Purcell et al., 2001), which also is not present in the assessment area (https://obis.org/). The harvest fish, *Peprilus alepidotus* (Harbison, 1993, GESAMP, 1997) and butterfish, *Peprilus triacanthus* (Oviatt and Kremer, 1977), are predators on *M. leidyi*, which appears to be nutritionally adequate for juvenile butterfish (https://www.cabi.org/isc/datasheet/75102). *Peprilus triacanthus* is present in Northern Labrador (https://obis.org/). *Cyanea* spp. are known to eat *M. leidyi* and negative correlations between *Cyanea capillata* and *M. leidyi* abundances have been reported from the York River estuary in Chesapeake Bay (Hosia and Titelman, 2010; and references therein). In the North Sea, the encounter rates between *C. capillata* and *M. leidyi* has been tested, and over 90% of encounters ended in escape, but the escaping *M. leidyi* suffered damage in the process (Hosia and Titelman, 2010). *Cyanea capillata* is native to the Canadian Arctic, and it has a circumpolar boreal northern distribution (https://obis.org/). Parasitic infections from the sea anemone *Edwardsiella lineata* are common and parasitized *Mnemiopsis* have significantly lower, sometimes negative growth rates in the native habitat. Prevalence can be higher than 50% and, as a consequence, parasite infection may be one of the factors that contribute to population control (Selander et al., 2010; and references therein). In the invaded range in the NE Atlantic, the occurrence of parasitic sea anemone larvae in *M. leidyi* was also reported (Selander et al., 2010). *Edwardsiella lineata* larvae may easily have survived transatlantic transport with ships ballast water, and arguably also by passive transport by the

Gulf Stream (Selander et al., 2010). No species from the *Edwardsiella* genus are known to occur in the assessment area (https://obis.org/).

From CABI: "A variety of fishes are known to consume gelatinous species, not only ctenophores (Harbison, 1993, Purcell et al., 2001). Some species more tolerant of temperate regions with low salinity were proposed for introduction into the Black Sea by the Gezamp group of experts (GESAMP, 1997). The Baltic cod, *Gadus morhua callarias* lives in the Baltic Sea near the bottom where the temperature is not higher than 14°C. The main food of cod is small pelagic fish and benthic animals. It is not yet known if cod can eat *M. leidyi*, however it eats other ctenophores, particularly *Beroe cucumis* (Kamshilov). Disadvantages are that it will also eat commercially valuable small pelagic fish. *Oncorhynchus keta*, the chum salmon, is an anadromous salmonid of high commercial value. It appears, in contrast to other representatives of the genus, to have gelatinous zooplankton as a major component of its diet. It spawns in the rivers and its early ontogenetic stages develop in fresh water, and is therefore not vulnerable to predation by *M. leidyi*. It is easily cultured, and its populations can be controlled in the rivers. However, it is not known if it eats *M. leidyi*." (https://www.cabi.org/isc/datasheet/75102)

Although not in the RA area, chum salmon are found in other areas of the Canadian Arctic.

Q7- Spread (potential natural dispersal) (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* has the capacity for a wide range of natural dispersal in the RA area. There is solid evidence of wide range natural dispersal.

Background Information: There is potential of transoceanic spreading of *M. leidyi*, as there are some specimen records in oceanic waters (Harbison et al., 1978), including localities inside the inflow of the Gulf Stream. Warming of the oceans may increase many populations of gelatinous species such as *M. leidyi* and also shift the population distributions poleward (Sullivan et al., 2001, Faasse and Bayha, 2006, Hansson, 2006). There is the potential of introduction through drift in the Baltic Sea (Haraldsson et al., 2013). In the HB LME, alongshore currents (Granskog et al., 2011) could disperse them counterclockwise from Churchill around the Hudson/James bay coasts and then east along the south coast of Hudson Strait. It is thought to have spread to the Sea of Marmara with the Black Sea currents (Shiganova, 1993).

Q8- Spread (potential anthropogenic dispersal) (Score = 3, Certainty = 2)

Score Rationale: Some reliable information suggests *Mnemiopsis leidyi* has the capacity for a wide range of anthropogenic dispersal in the RA area. There is solid evidence of widespread dispersal through ballast waters, though some uncertainty as to the degree of ballast currently being uptaken, transported and released within the RA area.

Background Information: It is hypothesized that *M. leidyi* was introduced by ballast water in the Black, Baltic, Mediterranean and North Seas (Oliveira, 2007, Purcell et al., 2007, Fuentes et al., 2010). Hence, it would be possible that the species is introduced to the Eastern Arctic by ballast water as well. There is a hypothesis of annual reintroduction by shipping for populations where the temperature drops 4ºC, like in certain parts of the North and Baltic Seas (Shiganova et al., 2001, Faasse and Bayha, 2006, Oliveira, 2007). Warming of the oceans may increase many populations of gelatinous species such as *M. leidyi* and also shift the population distributions poleward (Sullivan et al., 2001, Faasse and Bayha, 2006, Hansson, 2006). Ballast water transported by domestic coastal resupply and other boat traffic, and possibly hull fouling, could spread *M. leidyi* from Churchill north and west to coastal communities around western Hudson Bay, into Chesterfield Inlet, into northern Foxe Basin, and east to the Belcher Islands (Stewart and Howland, 2009, Chan et al., 2012). Chan et al. (2012) suggests that there is considerable domestic "Arctic direct" ballast exchange (about half of the total) in the region, suggesting that this is a plausible vector for within-region spread.

Q9- Impact on population (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* would have a high impact in many areas on population growth in the RA area. There is a considerable amount of reliable information on the high impacts of this species to populations and communities in its invaded range and similar effects would be expected in the RA area. It has been implicated in fishery declines.

Background Information: *Mnemiopsis leidyi* competes with commercial plankton feeding fish or preys on their juveniles, creating an economic impact as well (Korsun et al., 2012). In the assessment area there are several fish that are plankton feeders (Shorttail skate *Amblyraja jenseni*) (Stewart and Lockhart, 2005). It is known to be responsible for coastal environment deterioration, fishery declines and speeding up eutrophication related processes (e.g. anoxia, caused by massive precipitation of mucus and dead ctenophores to the bottom) (Mee, 1992, Volovik et al., 1993, Kideys, 1994, Shiganova, 1997). It directly competes with *Aurelia aurita* (native jellyfish in the Baltic). *Mnemiopsis leidyi* has a faster generation time and higher production rate and can outcompete *Aurelia* spp. as they occupy the same layer in the water column and compete for the same planktonic food items (Mutlu et al., 1994). There was a rapid decline in ichthyoplankton and mesozooplankton abundance and species diversity owing to predation of *M. leidyi* in the Black Sea (Shiganova, 1998). After its introduction in the Black Sea, there was a marked decrease in the biomass of non-gelatinous zooplankton (Kideys, 2002; and references therein). Although, in the North Sea it has been shown by field and lab experiments that *M. leidyi* has no serious potential as a direct predator of fish eggs, but individuals of this species might compete for food with larval fish as well as with the native ctenophore *Pleurobrachia pileus* (also native to the Canadian Arctic) (Hamer et al., 2010). Similar impacts would be expected on populations in the assessment area.

Q10- Impact on communities (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* would have high impacts in many areas on communities in the RA area. There is a considerable amount of reliable information on the high impacts of this species to communities in its invaded range and similar effects would be expected in the RA area.

Background Information: This species is a generalist carnivorous feeder. There were rapid declines in ichthyoplankton and mesozooplankton abundance and species diversity owing to predation from *M. leidyi* in the Black Sea (Shiganova, 1998). Changes in phytoplankton (can change from diatoms to dinoflagellates and cyanophytes), zooplankton, macrobenthos (change from domination by crustaceans (Corophidae, *Corophium robustum*) to annelids (*Nereis diversicolor*) and bivalves) and fish communities after its introduction to the southern Caspian Sea (Roohi et al., 2010). The same kind of impacts would be expected in the assessment area, since it is a generalist predator.

Q11- Impact on habitats (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* would have high impacts in many areas on habitat in the RA area. There are clear impacts on both the water column as well as benthic conditions.

Background Information: *Mnemiopsis leidyi* is a real ecosystem engineer. It affects physical conditions of several recipient productive ecosystems; for example, the decrease in water transparency, hydrochemical change, nutrient contents and biota (Shiganova, 2004). It is known to be responsible for coastal environmental deterioration, fishery declines and speeding up eutrophication related processes (e.g. anoxia, caused by massive precipitation of mucus and dead ctenophores to the bottom) (Mee, 1992, Volovik et al., 1993, Kideys, 1994, Shiganova, 1997).

Q12- Impact on ecosystem function (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* would have high impacts in many areas on ecosystem function in the RA area. There are clear impacts on both the water column as well as likely ones on the benthic ecosystem.

Background Information: After a *M. leidyi* invasion, cascading effects occurred at the higher trophic levels, from a decreasing zooplankton stock, to collapsing planktivorous fish, to dolphins (bottom up).

Similar effects occurred at lower trophic levels: from a decrease in zooplankton stock to an increase in phytoplankton, relaxed from zooplankton grazing pressure (top down) and from increasing bacterioplankton to increasing zooflagellates and infusoria (Shiganova, 2004). It can be predicted that increasing abundance of annelids and bivalves may lead to an increase in some benthopelagic fishes that feed on these benthic organisms (Roohi et al., 2010). *Mnemiopsis leidyi* occurrence and concentration, are reliable indicators not only reflecting on biodiversity, but also the ecological quality of the environment (Kamburska et al., 2006). It has caused a damaging impact on diversity and ecosystem stability through altering the Black Sea ecosystem structure and functioning, replacing indigenous species and provoking further introduction of aliens (*Beroe ovata* introduction in the late 1990s) (Kamburska et al., 2006). *Mnemiopsis leidyi* caused the collapse of the pelagic fish population in the Black Sea. It fed on the food supply of anchovies and on its eggs and larvae, and by eating herbivorous zooplankton caused an increase in phytoplankton biomass and primary productivity (Moncheva et al., 2001).

Q13- Associated diseases, parasites, and travellers (Score = 2, Certainty = 2)

Score Rationale: Some reliable information suggests *Mnemiopsis leidyi* would have high impacts in few areas or moderate impacts in many areas associated with diseases, parasites or travelers in the RA area. There is moderate information on parasites for *M. leidyi* elsewhere and a high probability of their transport with the host invasive. There is evidence that parasites of *M. leidyi* can use more than one type of species as a host and there are closely related potential host species in the RA area.

Background Information: Coelenterates are among the wide variety of planktonic and benthic invertebrates that may serve as intermediate hosts of fish parasites. Hysterothylacium larvae have been recorded in *M. leidyi* (Purcell and Arai, 2001; and references therein).

Parasitic infections from the sea anemone *Edwardsiella lineata* are common and parasitized *Mnemiopsis* spp. have significantly lower, sometimes negative growth rates in native habitats with prevalence of higher than 50% (Selander et al., 2010; and references therein). In the invaded range in the NE Atlantic, the occurrence of parasitic sea anemone larvae in *M. leidyi* was also reported (Selander et al., 2010). Selander et al. (2010) indicate that *E. lineata* larvae may easily have survived transatlantic transport with ship ballast water, and arguably also by passive transport in the Gulf Stream and could not exclude the possibility that *E. lineata* may parasitize some of the native ctenophores on the Swedish west coast where it has been found. No species from the *Edwardsiella* genus are known to occur in the assessment area (https://obis.org/). The species could potentially be transported with the *M. leidyi* host in ballast and could possibly infect native ctenophores known to occur in the assessment region, as it has been demonstrated to infect ctenophores of species other than *M. leidyi* (e.g., *Beroe ovata*) (Reitzel et al., 2007, Reitzel et al., 2009).

Q14- Genetic impact (Score = 1, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* would have low or no genetic impact on other species in the RA area. There are no related species of the same genus in the RA area.

Background Information: There are no native comb jellies in the same genus (https://obis.org/), so *M. leidyi* is unable to hybridize with local species.

Q15- Impact on at-risk species (Score = 2, Certainty = 1)

Score Rationale: Little to no reliable information is available to suggest *Mnemiopsis leidyi* would have high impacts in few areas or moderate impacts in many areas on at-risk or depleted species in the RA area. There are some species at risk that could be moderately impacted given the moderate effects of this ctenophore on ecosystems elsewhere. However, interactions and potential effects in the RA area are not known.

Background Information: There are no invertebrate or plant species at risk in Hudson Bay. As its invasion could bring cascading effects at higher trophic levels, species of fish at risk including northern wolffish, *Anarhichas denticulatus,* spotted wolffish, *Anarhichas minor,* and the thorny skate, *Amblyraja radiata* could be affected, and mammals at risk such as bowhead whale (*Balaena mysticetus*) could be affected since it feeds on small planktonic crustaceans (Stewart and Lockhart, 2005).

Q16- Impact on fisheries (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* would have high impacts in many areas on fished species in the RA area. There are clear impacts on fisheries species present in the water column, as well as benthic ones. There is an abundance of information on this species and its ecosystem effects from several invaded regions.

Background Information: Losses for the fisheries industry are large for countries surrounding the Black, Azov and Caspian Sea areas (https://www.cabi.org/isc/datasheet/75102). During the population explosions in the Black Sea in 1989 and 1995, anchovy fisheries collapsed in the region (Shiganova et al., 2001) causing an estimated loss of $250 million (NIMPIS 2002, from NOBANIS). It is expected to threaten sole, plaice and herring fisheries in the Baltic Sea (Faasse and Bayha, 2006, Oliveira, 2007). It has also been predicted to affect reproduction of cod in the Baltic because *M. leidyi* tend to aggregate near the halocline where Baltic cod spawning also occurs (Haslob et al., 2007). Competing for food with the main zooplanktivorous fish of the Caspian Sea, kilka (*Clupeonella* spp.), *M. leidyi* caused a dramatic recruitment failure of kilka from 2001 to 2004 resulting in a loss of at least 125 million US dollars to the Iranian economy (Roohi et al., 2010; and references therein). In the assessment area, harvest of subsistence fish and mammals could be affected due to potential cascading effects (bottom up).

Q17- Past invasion history (Score = 3, Certainty = 3)

Score Rationale: Considerable reliable information suggests *Mnemiopsis leidyi* is invasive elsewhere in the world. There is solid evidence of establishment and impacts in invaded areas.

Background Information: Native form the Atlantic coast of North and South America, *M. leidyi* first invaded the Black Sea in the 1980's through ballast water, where it spread to the Sea of Azov and the Mediterranean and Caspian seas (Purcell et al., 2007; and references therein); its recent appearance in the North Sea appears to be a separate introduction and has spread to the Baltic Sea (Purcell et al., 2007; and references therein). Thus, blooms may occur in areas where the species did not previously occur. Blooms in the Black Sea were possibly caused by climate change, eutrophication, fishing, construction and invasion (Purcell et al., 2007; and references therein). The species was introduced from the American coast to the Baltic Sea (Reusch et al., 2010) and is included in a list of 100 of the world's worst invaders (Lowe et al, 2000, from Siapatis et al., 2008). *Mnemiopsis leidyi* has also reached the southern North Sea (Boersma et al., 2007), and even the Wadden Sea (Tulp, 2006). In 2008 it was discovered that what had been identified as *M. leidyi* in northern Baltic waters was actually an Arctic species, *Mertensia ovum* (Fabricius, 1780), but because only larval and juvenile individuals had been found, the species was misidentified (Gorokhova et al., 2009). So far there have been no reports on occurrence of *M. leidyi* along the Eastern coastline of the Baltic Sea (Lithuania, Latvia, Estonia or Russia including the Kaliningrad).

CONCLUSIONS

Complete information to answer the questions related to the CMIST risk assessment tool for a total of 31 species for the HBC LME has been presented in this Data Report. Information gathered in this compilation of this set of species could be partially or totally used by other future risk assessments for regions other than the Hudson Bay Complex.

The information shown here was used as the basis of the analysis of two primary publications: Goldsmit et al. (2020) and Goldsmit et al. (2021). Refer to these to see further analysis on the predicted distribution of high-risk species identified or on the CMIST results and scores analysis, respectively.

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