Fisheries and Oceans Canada
Ecosystems and Oceans Science

Pêches et Océans Canada
Sciences des écosystèmes et des océans

Canadian Science Advisory Secretariat (CSAS)

Research Document 2020/003

National Capital Region

Effectiveness of Ballast Water Exchange Plus Treatment as a Mechanism to Reduce the Introduction and Establishment of Aquatic Invasive Species in Canadian Ports

D. Andrew R. Drake, Johanna N. Bradie, Dawson Ogilvie, Oscar Casas-Monroy, and Sarah A. Bailey

Great Lakes Laboratory for Fisheries and Aquatic Sciences
Fisheries and Oceans Canada
867 Lakeshore Road
Burlington, Ontario, L7S 1A1

February 2020

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca

ISSN 1919-5044

Correct citation for this publication:

Drake, D.A.R., Bradie, J.N., Ogilvie, D., Casas-Monroy, O., and Bailey, S.A. 2020. Effectiveness of Ballast Water Exchange plus Treatment as a Mechanism to Reduce the Introduction and Establishment of Aquatic Invasive Species in Canadian Ports. DFO Can. Sci. Advis. Sec. Res. Doc. 2020/003. viii + 60 p.

Aussi disponible en français:

Drake, D.A.R., Bradie, J.N., Ogilvie, D., Casas-Monroy, O., et Bailey, S.A. 2020. Efficacité de l'échange d'eau de ballast plus traitement comme mécanisme pour réduire l'introduction et l'établissement d'espèces aquatiques envahissantes dans les ports canadiens. Secr. can. de consult. sci. du MPO. Doc. de rech. 2020/003. viii + 65 p.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

The movement of ballast water is a prominent pathway for the dispersal of harmful aquatic species. As a continuous effort to better prevent invasions via this high-risk pathway, the current management strategy of ballast water exchange (BWE) will be gradually replaced by the International Maritime Organization's D-2 ballast water performance standard with the use of onboard ballast water management systems (BWMS). The Canadian Government proposed using BWE in concert with BWMS as this strategy may provide additional protection to certain ecosystems. Research on the performance of this strategy is required nationally and across different habitat types, so that informed decisions may be made on its implementation in Canada.

This study conducted a model-based analysis to estimate the invasion rate of non-indigenous zooplankton and harmful phytoplankton species via ballast water discharge in Canada under various ballast management strategies, with the objective to assess the relative performance of exchange plus treatment against exchange or treatment alone. Four management strategies were modelled: no management, BWE, ballast water treatment, and exchange plus treatment. Treatment was modelled by applying the D-2 standard on either all or half of the voyages to evaluate its effectiveness under different ballast water discharge compliance rates. These management scenarios were applied to five shipping pathways in Canada (i.e., Pacific International, Atlantic International, Great Lakes-St. Lawrence River (GLSLR) International, Arctic International, and Arctic Domestic). The management scenarios were also assessed on a port salinity basis, as environmental conditions such as salinity are known to influence the effectiveness of BWE.

The effectiveness of exchange plus treatment compared to treatment alone varied among shipping pathways and habitat types. With all vessels adhering to the D-2 standard, exchange plus treatment was the most effective management strategy at mitigating non-indigenous zooplankton establishments in GLSLR International and Arctic International, while exchange plus treatment did not provide additional invasion risk reduction for either taxonomic group over treatment alone for the other shipping pathways. For the source and recipient port salinity combinations, exchange plus treatment provided the greatest reduction in species establishment risk when the ballast source was either fresh or brackish water and the destination port was fresh water, while the efficacy of exchange plus treatment and treatment alone were similar for all other source and recipient port salinity combinations. When the D-2 standard was applied to only 50% of voyages, exchange plus treatment substantially decreased establishment risk when the ballast source was fresh water, regardless of the salinity of the recipient environment.

INTRODUCTION

Aquatic invasive species (AIS) are a prominent environmental stressor, having caused fundamental changes to Canada's aquatic ecosystems over the past 50 years. The ramifications of AIS can be severe, as their ecological impacts combined with other environmental stressors result in local and widespread biodiversity loss, including threatening the populations of species at risk and economically valuable species (Mills et al. 1993, Mack et al. 2000, Dextrase and Mandrak 2006). There are numerous pathways through which AIS are introduced beyond their native range, such as via commercial shipping (Casas-Monroy et al. 2014), recreational boating (Drake et al. 2017), and live trade industries (Marson et al. 2009, Bradie et al. 2013, Drake and Mandrak 2014).

Commercial shipping is a prominent invasion pathway, largely due to the movement of ballast water; most large commercial ships control their movement and stability by using ballast water (NRC 1996). During cargo unloading events, port water and, inadvertently, aquatic species are pumped into ballast tanks, which are then transported to regions that would otherwise not receive the organisms through natural dispersal. During cargo loading events, ballast water is discharged, releasing AIS into novel ecosystems. Some of the species that are taken aboard vessels may survive transit and, once released at the destination, a subset of those species may find favourable environmental conditions for survival and reproduction.

Significant ecological and economic impacts caused by ballast-mediated invasions have motivated governments and the global research community to understand the effectiveness of science-based management strategies in reducing the frequency and abundance of AIS arrival (i.e., propagule pressure). Considerable progress has been made to understand the role of ballast water as an invasion pathway, both globally and within Canada (Ruiz et al. 2007, Bailey et al. 2012, Casas-Monroy et al. 2014). Early indications suggest that the current ballast water exchange (BWE) strategy has reduced the rate of establishment of non-native species in low salinity Canadian ecosystems (Bailey et al. 2011, Bailey et al. 2012, Casas-Monroy et al. 2014). In recent years, onboard ballast water management systems (BWMS) have been developed, with the expectation that their use may achieve more consistent invasion risk reductions among different Canadian aquatic environments (Casas-Monroy et al. 2014). Although BWE and BWMS may substantially reduce the arrival of viable organisms to Canadian aquatic ecosystems, non-zero probabilities of arrival still exist (Cangelosi et al. 2011, Briski et al. 2013, Paolucci et al. 2015, Casas-Monroy et al. 2018). Therefore, the Canadian Government proposed that using BWMS in concert with BWE may ensure greater protection for certain aquatic ecosystems (IMO 2010). Although the limited research on this multidimensional approach to ballast water management has shown promising results under land-based and shipboard trials (Briski et al. 2013, Briski et al. 2015, Paolucci et al. 2015, Paolucci et al. 2017), additional research on its regional effectiveness is required prior to implementation in Canada.

BALLAST WATER EXCHANGE

Since the opening of the St. Lawrence Seaway in 1959 that allowed large transoceanic cargo vessels to travel up the St. Lawrence River to Great Lakes ports, ballast water discharge has been the primary pathway through which AIS were introduced to the Laurentian Great Lakes (Ricciardi 2006). Aiming to reduce invasion rates within this high priority pathway, the Government of Canada introduced ballast water exchange as a voluntary measure for vessels entering this region in 1989, which were expanded to all Canadian waters in 2000, then made mandatory in 2006 (Transport Canada 2007).

The Canadian regulations state that BWE must be performed ≥ 200 nautical miles from the nearest shore in water that is ≥ 2,000 metres deep, must achieve at least 95% efficiency of exchange and achieve a salinity level of ≥ 30ppt, and vessels entering the Great Lakes must "flush" empty ballast tanks with oceanic water to a specified standard (Transport Canada 2007). The 2006 Canadian BWE requirements vary depending on the geographical zone in which a ship operates (i.e., BWE exemptions zones), weather conditions, and vessel type (see Transport Canada 2007 for details). Alternate exchange zones located within Canada's economic exclusion zone (EEZ) allow for coastal BWE for certain domestic and international voyages (Transport Canada 2007).

Ballast water exchange is the process of discharging ballast water at sea, and filling ballast tanks with oceanic saline water in an effort to reduce the abundance of harmful coastal and/or freshwater species in ballast tanks. BWE utilizes environmental mismatching by exposing organisms to environmental conditions in which they may have a high probability of mortality. This is done via: 1) the purging of high-risk freshwater and coastal organisms in ballast tanks into mid-ocean water where their survivability is low (Ruiz et al. 2007, Reid 2012); 2) exposing residual organisms following exchange to large and abrupt changes in salinity which is lethal to most freshwater and some coastal species (Reid 2012); and, 3) the uptake of mid-ocean species during the exchange, with low probability of survival when discharged in recipient freshwater and coastal ecosystems (Reid 2012).

Since BWE relies on exploiting the effect of environmental fluctuations on organism fitness, the effectiveness of BWE varies among different habitat types. BWE is most effective at protecting freshwater environments as this is where the environmental mismatch is greatest (Casas-Monroy et al. 2014), especially when the ballast source is also freshwater (Santagata et al. 2008, Ellis and MacIsaac 2009, Bailey et al. 2011). BWE can reduce the concentration of freshwater organisms by up to 99.99% due to purging and osmotic shock (Bailey et al. 2011). Furthermore, there has been a reduced number of observed invasions in the Laurentian Great Lakes since both BWE and ballast tank flushing became mandatory in 2006, with only three reports of new species possibly introduced by ballast water since 2007 (Bailey et al. 2011, USEPA 2017a,b, Cangelosi et al. 2018). Relative to freshwater ecosystems, BWE is less effective at protecting coastal ecosystems from ballast-mediated invasions due to reduced environmental mismatching (McCollin et al. 2008, Cordell et al. 2009, Simard et al. 2011, Roy et al. 2012, Adams et al. 2014, Linley et al. 2014, Casas-Monroy et al. 2016). Mid-ocean species picked up during BWE – which in certain locations/seasons can be a relatively large number of species – have a higher probability of surviving in recipient coastal waters relative to freshwater ecosystems (Roy et al. 2012, Chan et al. 2015, Casas-Monroy et al. 2016). Furthermore, when the port of origin is of high salinity, any residual high-risk species not purged during BWE are more likely to tolerate the salinity of mid-ocean water following BWE and may benefit from the renewal of oxygen in the ballast tanks (Reid 2012, Bailey 2015). Due to the abovementioned limitations of BWE, it has been considered a short-term solution until more effective management strategies such as ballast water discharge performance standards could be employed (IMO 2018).

BALLAST WATER MANAGEMENT SYSTEMS

With the functional aim to reduce the risk of invasions by decreasing propagule pressure of discharged AIS, the International Maritime Organization's (IMO) International Convention for the Control and Management of Ships' Ballast Water and Sediments (hereafter referred to as the Convention) – which was adopted in 2004 and entered into force on September 8, 2017 – sets limits on the concentration of organisms that can be discharged in ballast water, i.e., Regulation D-2 (see Table 1 for details; IMO 2018). The Convention only applies to ships operating across international jurisdictions (excludes domestic shipping, unless the discharge of such ballast water is determined to impair or damage the environment, human health, property or resources), and it will be gradually implemented over time depending on the size and age of the vessel (IMO 2004). In the meantime, vessels are expected to perform BWE following Regulation D-1. Parties to the Convention are expected to adhere to the regulations set by the Convention and they retain the ability to implement stricter regulations in their respective countries to provide superior protection to their aquatic environments (IMO 2004).

Shipboard BWMS are considered the most feasible method to adhere to the D-2 standard; BWE alone cannot achieve these standards when considering the uptake of oceanic organisms during exchange, and BWE may not be logistically feasible for coastal voyages due to time and distance-from-shore constraints (Bailey et al. 2011, Bailey 2015). BWMS utilize wastewater treatment technologies to reduce the concentration of organisms in ballast tanks, typically through a combination of disinfection (e.g., ultraviolet radiation, electrolysis, chemical injection), and filtration (e.g., screen or disc filters, hydrocyclones) processes (Mouawad Consulting 2013). Certain types of disinfection processes may only be used during ballast water uptake before the water enters ballast tanks (e.g., electrolysis), while some processes may also treat ballast water during discharge (e.g., ultraviolet radiation; Mouawad Consulting 2013).

At least 76 BWMS have received certification to date, demonstrating these BWMS can treat ballast water to the organism discharge limits as defined by the D-2 standard, at least under specified test conditions (see IMO 2019 for details). Additionally, studies have determined that a variety of ballast water treatment technologies can significantly reduce the abundance of aquatic organisms (Gregg et al. 2009, Casas-Monroy et al. 2018), indicating that BWMS could be effective at mitigating ballast water mediated invasions. However, non-zero probabilities of species establishment still exist, as some BWMS may malfunction, be maintained or operated incorrectly, or be utilized in challenging waters outside of the limiting operating conditions (e.g. very high turbidity), which may result in a failure of the ballast water to meet the D-2 standard. In addition, some species may be resistant to certain treatments: Briski et al. (2015) determined that Copepoda may survive ballast water treatment, and de Lafontaine et al. (2008) concluded that chemical treatments may be resisted by nematodes that bury in sediment and Zebra Mussels (*Dreissena polymorpha*) that close their shells when exposed to toxic chemicals. High turbidity has been shown to reduce the efficacy of ultraviolet radiation treatments (Briski et al. 2013), and large quantities of filamentous algae can clog filtration systems (Cangelosi et al. 2011). This is problematic as many fresh and brackish ports have turbid water when located near rivers, making turbidity a frequently encountered issue (Briski et al. 2013). As a result, failures to meet the D-2 standard may occur regularly during the early years of the Convention. However, the performance of BWMS is expected to improve in the future with advancements in ballast water treatment technologies as experience with their use is gained.

EXCHANGE PLUS TREATMENT

To protect Canada’s aquatic systems, Canada reserves the right to introduce stronger regulations than those set by the Convention (IMO 2004). Given this reservation of authority, in 2010, the Government of Canada proposed to the IMO that combining exchange with treatment may provide greater protection against invasion than BWMS could alone for fresh and brackish water ports (IMO 2010). The mechanism of action of this combined strategy is to target multiple components of the invasion process by reducing the probability of organism survival through environmental mismatching through BWE, and reducing propagule pressure through ballast water treatment. Furthermore, BWE can serve as a backup strategy in the event that BWMS malfunction.

The two protocols for ships to conduct exchange plus treatment are treatment plus exchange plus treatment (T+E+T) and exchange plus treatment (E+T). All water is managed using BWMS before entering ballast tanks for the T+E+T method, while for the E+T method, the BWMS is bypassed during the initial ballast uptake at port, then used during BWE to treat incoming oceanic water. Under less challenging port conditions, outside of seasonal challenges such as water turbidity that worsens during spring flooding events, T+E+T may provide superior protection against invasions compared to E+T because all water entering ballast tanks is managed, and BWMS can serve as a backup strategy for when it is unsafe to perform BWE due to poor weather conditions (Briski et al. 2013). Furthermore, T+E+T meets current IMO regulations as the discharge of untreated ballast water at sea is not permitted (IMO 2004). The benefit of E+T is that it places less stress on BWMS during challenging port uptake conditions that may otherwise cause the BWMS to go offline for repairs and maintenance (for example, due to clogged filters), and also has lower associated effort and cost due to fewer required treatment steps (C. Wiley, past Chair of IMO Ballast Water Review and Working Groups, pers. comm., Briski et al. 2013). The drawback of E+T is that port water is only managed using BWE, where the critical salinity barrier is absent when either the source or recipient port is of high salinity. This may be a serious concern as residual ballast in a ship following BWE may still contain a substantial amount of viable plankton, posing a risk of invasion to recipient ecosystems (Duggan et al. 2005. Duggan et al. 2006). Additionally, discharging untreated port water at sea may release potentially viable invasive species into these environments, presenting a risk to mid-ocean ecosystems.

In recent years, scientific research comparing the effectiveness of exchange plus treatment versus treatment alone under land-based and shipboard trials determined that exchange plus treatment may provide greater protection against invasive species, especially when used between freshwater source and recipient ecosystems. Briski et al. (2013) concluded that exchange plus treatment reduced the concentration of zooplankton and phytoplankton greater than treatment alone, when only freshwater organisms are considered. Furthermore, Briski et al. (2015) determined that when exchange plus treatment is used between freshwater ecosystems, discharged ballast water mainly contained lower risk marine species, while ballast water managed using BWMS alone primarily discharged higher risk freshwater and euryhaline species. Additional studies have demonstrated that exchange plus treatment may reduce both propagule and colonization pressures greater than treatment alone, when all organisms are considered (Paolucci et al. 2015, Paolucci et al. 2017). These studies demonstrate the benefits of exchange plus treatment, especially its potential to reduce the abundance of freshwater and euryhaline species in ballast water, which have a higher risk of establishing in freshwater ecosystems. However, the effect of exchange plus treatment compared to that of BWMS alone on the probability of AIS establishment as applied to specific habitat types (i.e., fresh, brackish, or marine) or to geographical regions containing diverse aquatic habitats must be studied further.

OBJECTIVE

The objective of this study was to conduct a model-based analysis of biological and shipping data to estimate the expected AIS establishment rate in Canada attributed to ballast water discharge, under different management scenarios. The focus was on the effectiveness of exchange plus treatment compared to that of either exchange or treatment individually, as it was important to determine if this alternative management strategy provides greater reductions in invasion risk. This study also considered the possibility that ballast water may not meet the D-2 standard consistently during the early years of BWMS use and determined the effect of using BWE as a backup strategy.

METHODS

STUDY AREA

The analysis of Canadian ecosystems incorporated six ballast water management scenarios, namely no-management, exchange-only, treatment-only (100%), treatment-only (50%), exchange plus treatment (100%), and exchange plus treatment (50%); see Table 2 and Modelling the Management Scenarios section for details. These management scenarios were applied to five combinations of Canadian geographical regions – the Pacific, Atlantic, Arctic, and Great Lakes-St. Lawrence River (GLSLR) – and shipping traffic pathways – international or domestic – with each combination henceforth referenced as a shipping pathway (see Table 3 and Figure 1 for details). The international pathway, which was evaluated for all four Canadian regions, included all vessels entering Canada's jurisdiction from foreign ports (excluding vessels transiting between the American and Canadian GLSLR ports), and the domestic pathway only considered ships arriving to an Arctic port from another Canadian region (Table 3). The domestic pathway was only considered for the Arctic because alternate exchange zones enable the action of BWE for ships entering this region, whereas for ships travelling between other Canadian regions, BWE is operationally infeasible due to limited distance-from-shore. Additionally, extra prevention may be warranted for the Arctic region, as the Arctic is a unique bioregion containing several ecologically sensitive areas, some of which are endemic to the region (Arctic Council 2009, Chan et al. 2012). Lastly, to assess the extent of which the management strategies would affect Canada as a whole, the shipping pathways were combined to generate an All Shipping Pathways option.

Since the effectiveness of BWE is influenced by differences in salinity, each management scenario applied to each shipping pathway was estimated in relation to various source and recipient port salinity combinations, based on three salinity categories (fresh, brackish, and marine). The thresholds for the salinity categories were ≤ 5.0‰, 5.1-18.0‰, and ≥ 18.1‰ for fresh, brackish, and marine water, respectively, following the changes in species richness across a salinity gradient as described by Remane and Schlieper (1972).

This model did not evaluate species-specific risk. Instead, invasion rates are based on the taxonomic groups of zooplankton and phytoplankton which were modelled separately. Due to a lack of data regarding which zooplankton species are harmful and which phytoplankton are non-indigenous species, this study assessed the species establishment rates of non-indigenous zooplankton and harmful phytoplankton attributed to ballast water discharge. Henceforth, non-indigenous zooplankton and harmful phytoplankton are collectively referred to as harmful species or harmful individuals.

SHIPPING AND BIOLOGICAL DATA SOURCES

This quantitative assessment used much of the peer-reviewed biological and shipping data previously obtained for Fisheries and Oceans Canada's National Risk Assessment for the Introduction of Aquatic Nonindigenous Species to Canada by Ballast Water (Casas-Monroy et al. 2014). This included biological data from the Canadian Aquatic Invasive Species Network (CAISN) – Humphrey (2008), Klein et al. (2009), Bailey et al. (2011), Briski et al. (2012a,b), Casas-Monroy (2012), DiBacco et al. (2012), Roy et al. (2012), and Adebayo et al. (2014) – which has undergone extensive peer review by working groups of regional experts, academics, and government scientists and is of high scientific quality. The biological data can be made available on request. The shipping traffic data used for the GLSLR International, Pacific International, and Atlantic International pathways were obtained from Casas-Monroy et al. (2014) and incorporated the following databases: Transport Canada Ballast Water Database

(TCBWD), Canadian Coast Guard's Information System on Marine Navigation, and the U.S. National Ballast Information Clearinghouse.

Data from Casas-Monroy et al. (2014) for the Arctic region was updated for this study, using zooplankton biological data from Chan et al. (2015) for the Arctic International pathway, and additional Arctic shipping data from TCBWD, Canadian Coast Guard Northern Canada Vessel Traffic Services, and Fednav Inc. Due to the unavailability of phytoplankton biological data for Arctic International and Arctic Domestic, it was assumed that the phytoplankton concentration and composition was equivalent to that of Atlantic International. Additionally, zooplankton data was unavailable for domestic voyages arriving to the Arctic from the Great Lakes and, therefore, relevant data from internal GLSLR transits were used for the Arctic Domestic pathway. The environmental data (i.e., salinity and temperature) used for the survival component of the model was obtained from Keller et al. (2011) and World Ocean Atlas 2013 Vol. 2 (Locarnini et al. 2013, Zweng et al. 2013); the modelled salinities of inland freshwater ports were corrected where necessary. The Canadian ports and their salinity values used in this study are provided in Appendix 1.

MECHANISTIC MODEL-BASED APPROACH TO ESTIMATE THE SPECIES ESTABLISHMENT RATE IN CANADA

This analysis incorporates a mechanistic model to evaluate the relative performance of the ballast water management strategies of interest. The mechanistic, model-based approach is advantageous because: 1) it does not incorporate proxy variables of organism concentrations, so the actual discharge concentrations and their changes in response to ballast water management are modelled explicitly for a given baseline invasion rate; 2) it is not influenced by the many issues associated with species discovery data (NRC 2011, Wonham et al. 2013); and, 3) mechanistic approaches work well when extrapolating beyond observed conditions (Bolker 2008) as is required to estimate the change in invasion probability associated with ballast water management.

To estimate the number of harmful species establishing in Canadian ecosystems annually, agent-based simulations involving three main components were conducted: 1) the number and concentration (e.g., individuals/m^3) of harmful species discharged to Canadian ecosystems; 2) the survival probability of these species based on the environmental correspondence between ballast source and recipient locations; and, 3) species establishment based on their initial population sizes and per-capita establishment probabilities. Appendix 2 outlines the values of the input parameters used in the model.

SIMULATION OF SHIPPING ACTIVITY

The first step in the analysis was to construct a year-long iteration of shipping activity for each shipping pathway based on the defined shipping data, which described the origin to destination movement of ballast water on individual voyages to Canadian ecosystems (see Appendix 2 for years of observed shipping activity and Table 4 for ship-trip sample size). Incorporating a one-year iteration of shipping activity allowed differences in the frequency of vessel transits and their effect on invasion risk to be standardized among the different temporal scales across each shipping pathway. Obtaining the number and spatial trajectory of voyages was necessary to determine the number of release events within a given time period for each shipping pathway and the environmental correspondence between a source and recipient port for a given transit.

ESTIMATING SPECIES ARRIVAL

The second component of the model – which consisted of four steps – used the empirical biological data to estimate the number of harmful species and their initial population sizes discharged into Canadian ecosystems per trip. Quantifying initial population size was necessary as it is one of the parameters used to determine the probability of species establishment. Estimations were conducted separately for zooplankton and phytoplankton and each shipping pathway of interest.

First, sample organism densities were obtained from empirical ballast tank samples (e.g., samples of zooplankton obtained during a single net haul), which included both juvenile and adult individuals as well as both harmful and benign (or non-indigenous and indigenous) species in the total organism count. These empirical sample organism densities among ship-trips were plotted, and maximum likelihood – a common statistical technique in biological research – was used to estimate the most likely statistical distribution of total organism density (individuals/m^3 for zooplankton and cells/mL for phytoplankton), given the empirical data (see Appendix 2 for specific values). Based on an Akaike Information Criterion (AIC) statistic, the variation among biological samples was best described by a negative binomial statistical distribution with parameters, size (e.g., dispersion) and μ (mean; Casas-Monroy et al. 2014). Essentially, this step generated a probability distribution for each pathway characterizing the total density of zooplankton or phytoplankton in ballast tank samples (Figures 2 and 3). For the Arctic domestic pathway, separate distributions were generated for arrivals from the Atlantic and GLSLR regions (the latter using data from internal GLSLR transits as indicated above); arrivals from the Pacific were not considered since there was no shipping activity to the Arctic originating in the Pacific.

Second, since the probability distributions generated above are based on the concentration of organisms in ballast tank samples, a conditional parameter of the population density given the sample density ($D_p \mid D_s$) was incorporated in simulated data to estimate the statistical population density of organisms in a given tank (Figure 4). Estimating the statistical population density was necessary as sampling error may cause the sample density to deviate from the actual population density of organisms inside a ballast tank. With the conditional parameter, it was assumed that the underlying spatial distribution of organisms in a ship ballast tank are distributed according to a Poisson process, such that when a given sample density is obtained, the population density of that ship's tank may be slightly higher or lower than that of the sample. This is similar to the approach used by Lee et al. (2013) that acknowledges a given concentration standard may be exceeded even when a single sample is below the value of interest. A summary of the approach to estimate the statistical distribution of propagule pressure is outlined in Drake et al. (2015).

Third, since the population density in a tank may contain both indigenous and non-indigenous (or harmful and benign) individuals, the proportion of non-indigenous zooplankton or harmful phytoplankton individuals out of the total population was estimated (Figures 5 and 6). This was done by summarizing the fraction of organisms in empirical samples that were non-native zooplankton or harmful phytoplankton for each shipping pathway; this empirically derived approach incorporated the biogeographic context of species arriving to different geographic regions (e.g., one species may be native to Atlantic Canada while non-native to Pacific Canada; Casas-Monroy et al. 2014). Although both juveniles and adults were included in the sample density, only adults identified to the species level were considered in the fraction of non-indigenous organisms. It was assumed that the proportion of non-indigenous adults was equivalent to the proportion of non-indigenous juveniles. A beta distribution was fit to the proportion of harmful individuals for each geographic region using maximum likelihood, allowing the relative proportion of harmful individuals transported to be specific to each shipping pathway (see Appendix 2 for shape parameters α and β). Beta distributions are continuous probability distributions (range 0 through 1) and can be used to reflect the distribution of proportions, as would be expected across a range of ballast tanks with different relative abundances of harmful individuals in a given shipping sector. Essentially, this step generated a probability distribution for each pathway characterizing the proportion of non-indigenous zooplankton or harmful phytoplankton organisms in ballast tank samples.

Fourth, since multiple species exist in a single discharge event, it was important to not only characterize the total number of harmful individuals on a given ship-trip, but also to estimate the underlying species abundance distribution (i.e., the distribution of *n* individuals among *n* species; Drake et al. 2014). Multiple species abundance distributions were available for each pathway based on the pathway-specific empirical biological data, determining the total number of non-indigenous or harmful species and their relative proportions out of the total abundance of non-indigenous/harmful individuals on each voyage (see Table 5 for the number of species abundance distributions for each shipping pathway).

In summary, for each shipping event, a value was randomly drawn from the sample density distribution to determine the sample density of organisms. For each sample density, a random value was selected from the $D_p \mid D_s$ distribution to determine the population density of organisms. Next, a random draw was made from the non-indigenous/harmful distribution, representing the fraction of the population that is harmful or non-indigenous for that trip. These steps determined the total number of harmful individuals discharged for a given ship-trip (i.e., total density of individuals * proportion of harmful individuals). Then, a random species abundance distribution was selected, determining the number of harmful species and their respective abundances out of the total number of harmful individuals.

ESTIMATING SPECIES SURVIVAL

After the number of harmful species and their concentrations were estimated for a trip, it was then determined whether those species would survive the environmental conditions following discharge into a recipient port based on the environmental correspondence between the port of origin (or location of exchange, if applied management included exchange) and destination ecosystem. Survival probability was needed both to reflect the ecological effect of BWE, as well as the effect of environmental matching among source and recipient ports when BWE is not conducted. Note that the term survival is used here to describe initial survival of a species in new environmental conditions, rather than the long-term suitability of the environment for a species.

Water temperature (mean, maximum, and minimum) and salinity data were used to calculate the environmental distance between ballast origin and destination environments, as these are broad-scale variables that influence the distribution and survival of aquatic species (Casas-Monroy et al. 2014). Environmental distance was calculated as the Euclidean distance between standardized variables following Bradie et al. (2015). To determine the relationship between environmental distance and the probability of survival, a binomial generalized-linear model was fit with presence-presence distances versus presence-background distances using data for 603 aquatic species – from the kingdoms of Animalia and Plantae – that have invaded one or more regions. Presence data was obtained from the Global Invasive Species Information Network (species with less than three unique occurrence points excluded; GISIN 2014). The species dataset was split into 80% for fitting and 20% for training, and the area under the curve (AUC) was used to evaluate model performance (AUC = 0.94). When the temperature and salinity of the source and recipient ports matched, the probability of survival was high (Figure 7). The environmental distance model reflects the average probability of survival for propagules of a species given the environmental match between ports.

Once the environment match was determined, a random draw was made from the logistic function to determine whether each of the *n* species survived the recipient port conditions, with surviving species moving on to the establishment component in the model. Note that this study only considered the survival or mortality of species, and not the fitness of those species that survived the recipient environmental conditions or effects of treatment.

ESTIMATING ESTABLISHMENT

The introduced species that survived the environmental conditions of the novel ecosystem were assessed to determine whether they would establish viable populations. First, the establishment probability (1 – the probability of extinction) of each surviving species was determined. This analysis incorporated the probabilistic establishment model from the National Research Council (2011; Box 4-1, Equation 4-3; see also Leung et al. 2004, Jerde et al. 2009) that describes the propagule pressure-establishment relationship using the equation $P_e = 1 - e^{-\alpha N^c}$; where P_e is the probability of establishment, α is the probability that a single individual will establish a viable population (i.e., a species' per-capita establishment probability), N is the initial population size, and c describes the existence of an Allee effect ($c > 1$). It was assumed that an Allee effect was not present (i.e., $c = 1$; Bradie et al., 2013), allowing for the realistic establishment of parthenogenetic species when initial population sizes are small, and giving a hyperbolic shape to the establishment curve (Leung et al. 2004). Examples of establishment probability curves are presented in Figure 8; when α is large, the probability of establishment is high even when the initial population size is low, unlike for small α values, where larger initial population sizes are required to reach high probabilities of establishment.

The α parameter is expected to vary by species and true α values are unknown. Therefore, a beta distribution with shape parameters, $\alpha = 0.005$ and $\beta = 5$, was assumed to describe the distribution of α values across multiple species in a ballast tank. This distribution was designed to include a wide range of aquatic species and their per-capita establishment probabilities under diverse biological, chemical, and physical conditions, and the distribution was assumed to be identical for both non-indigenous zooplankton and harmful phytoplankton as well as across all voyages and shipping pathways. The distribution of α was chosen with the understanding that in general, most species have very low α values, while the 'worst-case' parthenogenetic species are expected to have the highest α values. An appropriate α value for parthenogenetic species was informed by data from Bailey et al. (2009), detailing the upper-limit of establishment probability for parthenogenetic zooplankton in the Great Lakes. Using this data, the distribution was fit to align the upper-limit of α for parthenogenetic species to the 97.5th percentile of the curve, with the majority of species having α values well below this value (Figure 9); marginally lower α values would be expected for less successful asexual species and much lower α values would be expected for sexual species. The use of this distribution was empirically supported by the low proportion of parthenogenetic species found in biological samples from Casas-Monroy et al. (2014). Additionally, the mean α parameter for this curve is equivalent to the mean α parameter estimated for empirical data for establishment of non-indigenous fishes (Bradie et al. 2013), further supporting the utility of this curve for representing real-world distribution of α values among species. Details on the uncertainty in this component of the model are provided in Sources of Uncertainty and Considerations for Re-Running the Model sections.

Thus, for each shipping event, an α value was randomly selected from the α distribution for each surviving species, and their probability of establishment was estimated using corresponding α values and initial population sizes. Then, the establishment (1) or extinction (0) of each species was determined using a Bernoulli trial based on their probability of establishment.

The estimation of species establishment per trip and among all yearly trips within each shipping pathway allowed for the retention of two metrics of species establishment risk: 1) the number of harmful species establishing per year, and 2) the probability that at least one species establishes per trip. The number of species invasions per year represents the number of species establishments annually, when considering the one-year iteration of shipping traffic. The probability that at least one species invasion occurs per trip reflects the species establishment risk of individual trips and is largely independent of shipping traffic volume. For ease of interpreting the results, the annual number of species establishments per year was multiplied by ten to result in the number of species invasions per decade.

MODELLING THE MANAGEMENT SCENARIOS

A key assumption of the model was that the biological distributions for estimating the number and abundance of each harmful species per voyage were identical for no-management and BWE. This assumption was necessary because the available data was based on empirical samples of ships presently conducting exchange. While most studies have showed an influence of exchange on organism abundance (Wonham et al. 2001, Gray et al. 2007, Simard et al. 2011), others have shown much greater variability, especially in marine systems (Ruiz and Smith 2005), and evidence from CAISN sampling and Chan et al. (2015) indicates that total concentration of organisms can remain similar between control and exchanged tanks. Therefore, the modelled effect of BWE was only a change in organism survival probability reflective of the environmental distance between the locations of exchange and recipient ports, and only mid-ocean species were modelled following BWE. The approach of assuming no change in population density, but rather an effect of environmental mismatching, was pursued because it introduced the fewest assumptions and provided the greatest data availability to inform the region-specific parameters. It is recognized that the complete effects of BWE may not be modelled in this study, as BWE may affect the total organism abundance, and may not remove 100% of source port organisms, resulting in residual euryhaline and marine species possibly having better survivability than modelled; these factors influence the relative species establishment rates under scenarios with BWE compared to those without BWE, although they are beyond the scope of this study. The exchange location was randomly selected from population data of the actual mid-points of exchange for ships arriving to each geographic region of Canada in 2015.

Treatment was modelled by manipulating the total population density of organisms of the no-management scenario according to levels imposed by the D-2 standard (i.e., the population density was reduced to 10 zooplankton individuals/m^3 and 10 phytoplankton cells/mL) on either all or half of voyages. The population density was unchanged if it was already below the D-2 standard. Exchange plus treatment was modelled by applying the abovementioned effects of treatment to the exchange-only scenario, also on either all or half of voyages. The example of 50% of voyages was chosen based on unpublished data of treated ballast water samples collected during April 2017- December 2018; it is expected that the reliability of BWMS will improve in the future with increased operational experience and advancements in treatment technologies.

MODEL SIMULATIONS

As the relative identity of each harmful species was retained when drawing from the species establishment distribution, multiple establishment events of a single species were recorded but tallied as a single invasion. This was necessary to avoid double-counting establishment events, given that the statistic of interest is the number of unique species establishments. The entire resampling process was repeated 1000 times (i.e., 1000 yearly iterations of shipping activity for each shipping pathway), and the mean and bootstrapped 95% confidence interval were catalogued for both metrics of invasion risk within each shipping pathway. The 95% confidence interval was calculated by bootstrapping the annual results with replacement 5000 times to determine the 95% confidence interval of the mean. The statistic of interest was the expected value, representing the expected number of species over a long-run average, which is the weighted average of all outcomes by their relative probability, and is a continuous value (e.g., 2.1 species expected per year).

MODEL CALIBRATION AND RESULT STANDARDIZATION

Since the Laurentian Great Lakes was the only broad Canadian region with sufficient AIS discovery data, the model output across all shipping pathways was calibrated concurrently by comparing the modelled invasion rates of no-management and exchange-only for GLSLR International to the ship-mediated species arrival estimates in the Great Lakes from Ricciardi (2006).

The percentage change in species per decade and probability of species establishment per trip of each of the management scenarios, compared to the no-management or exchange-only baselines, was calculated for both non-indigenous zooplankton and harmful phytoplankton. The relative differences between management scenarios within a given shipping pathway should be prioritized when interpreting the model results, due to the high degree of uncertainty associated with predicting absolute species establishment rates given current scientific knowledge of invasive species. Since the propagule pressure-establishment relationship is highly context-specific for biotic and abiotic variables not included in this study (e.g., habitat suitability beyond temperature and salinity) and is unknown for the vast majority of species, the approach used here is supported by ecological theory but has not been validated experimentally. Conducting an absolute assessment could involve undertaking hundreds of population viability experiments for a wide variety of species, each in different recipient environments. Without such empirical data, a high level of uncertainty would result. However, the relative effectiveness among the management scenarios is likely to remain constant even though the actual values defining the propagule pressure-establishment relationship are unknown, or if the values used to define this relationship change as scientific knowledge on the subject advances and lead to different absolute establishment rates than those of this study.

SENSITIVITY ANALYSIS

Sensitivity analyses were run to examine how model results would change in response to deviations in the model parameters. Model sensitivity was evaluated in two parts, (1) parameters that could alter the overall expected number of invasions, but not the relative performance of management methods, and (2) parameters that could alter the relative performance of management methods. For part 1, a 25% shift (increase and decrease) was applied to the transit frequency, mean sample density μ, and mean proportion of harmful or non-indigenous individuals β. In addition, the source and recipient ports within each shipping pathway were randomized to determine the effect of changing trade patterns within a given geographic sector on model results. The parameters α and c in the establishment equation were set to 0.05 and 2, respectively, for all species in the abovementioned trials. For the second part of the sensitivity analysis, the modelled effect of BWMS in the treatment-only (50%) scenario was altered by applying post-treatment organism concentrations equivalent to those observed in preliminary evaluations of treatment efficacy (Casas-Monroy and Bailey, Fisheries and Oceans Canada, unpublished data). The 50% 'pass' rates (i.e., ships with ballast that meets the D-2 standard) for non-indigenous zooplankton was maintained to match the field data, but post-treatment data for both 'pass' and 'fail' events was produced from probability distributions generated from real 'pass’ and ‘fail’ events. Therefore, with this approach, treatment could be partially effective on the 50% of voyages where ballast did not meet the D-2 standard by allowing the population density to be reduced to a value above the standard and, for the 50% of voyages that adhered to the D-2 standard, treatment could reduce the population density to below the standard (i.e., 0-10 individuals per m^3). This portion of the sensitivity analyses was not run for harmful phytoplankton as observed efficacy was 100%, which is equivalent to the modelled scenario of treatment-only (100%).

RESULTS

Results are presented as the expected number of harmful species establishing in aquatic ecosystems in Canada. The significance of different establishment rates is not discussed, as this is dependent on the risk tolerance of policy makers and risk managers. Note that the model results involving the application of the D-2 standard only reflect the expected establishment rates for non-indigenous zooplankton and harmful phytoplankton; it is possible that additional establishments may occur from groups of taxa not modelled (e.g., native zooplankton; non-indigenous but non-harmful phytoplankton).

ALL SHIPPING PATHWAYS: SPECIES PER YEAR

When all shipping pathways were combined, treatment-only (100%) reduced the overall expected number of species invasions per decade (SpPD) of non-indigenous zooplankton from 21.61 (no-management) to 3.27 (Figure 10). This was a considerable improvement to that achieved using BWE, where 21.68 non-indigenous zooplankton species were expected to become established per decade. The trends in the results were mirrored for harmful phytoplankton, where the application of the D-2 standard on all ship-trips (4.79 SpPD) greatly outperformed BWE (16.46 SpPD), when compared to no-management (16.25 SpPD; Figure 11). The model results for the probability of species establishment per trip for each shipping pathway is available in Appendix 3.

PACIFIC INTERNATIONAL PATHWAY

For the Pacific International pathway, BWE had similar establishment rates to those of no-management for both non-indigenous zooplankton (exchange-only, 12.69 SpPD vs. no-management, 12.69 SpPD) and harmful phytoplankton (exchange-only, 11.11 SpPD vs. no-management, 11.17 SpPD; Figures 10 and 11). Moving from BWE alone, treatment-only (100%) lowered SpPD to 1.51 for zooplankton (88% reduction) and to 2.75 for phytoplankton (75% reduction; Figures 10-12); similar risk reductions resulted from using exchange and treatment in concert (1.6 SpPD for zooplankton and 2.57 SpPD for phytoplankton). When the D-2 standard was only applied on 50% of ship-trips, the effectiveness of both treatment alone and exchange plus treatment were similarly reduced, resulting in an SpPD of 10.49 (treatment) and 10.58 (exchange plus treatment) for zooplankton, and 9.38 (treatment) and 9.67 (exchange plus treatment) for phytoplankton.

ATLANTIC INTERNATIONAL PATHWAY

When ballast water was unmanaged, 2.22 non-indigenous zooplankton and 12.74 harmful phytoplankton species were expected to establish per decade via the Atlantic International pathway, while establishment rates were largely unaffected by the use of BWE (zooplankton, 2.31 SpPD; phytoplankton, 12.65 SpPD; Figures 10 and 11). In comparison to BWE, treatment (50%) had lower invasion rates for both zooplankton (1.96 SpPD) and phytoplankton (11.04 SpPD). However, species establishment rate was greatly reduced for both zooplankton (0.13

SpPD) and phytoplankton (3.07 SpPD) when all ships' ballast water were treated to the D-2 standard. In both instances, very little difference in outcomes were observed by using treatment-only as opposed to exchange plus treatment; exchange plus treatment (50%) had an SpPD of 1.92 (zooplankton) and 11.14 (phytoplankton), and exchange plus treatment (100%) had an SpPD of 0.22 (zooplankton) and 3.26 (phytoplankton).

GREAT LAKES-ST. LAWRENCE RIVER INTERNATIONAL PATHWAY

For the GLSLR International pathway, BWE was slightly more effective at reducing species establishment risk compared to unmanaged ballast water; SpPD for zooplankton decreased from 5.95 (no-management) to 5.14 (BWE), and SpPD for phytoplankton was lowered from 0.87 (no-management) to 0.74 (BWE; Figures 10 and 11). Exchange-only was equally as effective as treatment-only (50%) for phytoplankton, but SpPD for zooplankton decreased 12% to 4.54 under treatment alone (Figures 10-12). Combining exchange and treatment (50%) produced greater species establishment risk reductions than achieved by each on its own, especially for harmful phytoplankton; relative to BWE alone, exchange plus treatment (50%) reduced invasion rates by 20% to 0.59 SpPD (harmful phytoplankton) and 19% to 4.15 SpPD (non-indigenous zooplankton). The efficacy of BWMS drastically improved when all vessels adhered to the D-2 standard, which resulted in an SpPD of 1.39 for zooplankton (73% decrease from BWE) and 0.25 for phytoplankton (66% decrease from BWE). Compared to treatment (100%), the addition of BWE with treatment had an identical SpPD for harmful phytoplankton, and a slightly lower SpPD of 1.1 for zooplankton (79% reduction from BWE alone).

For the source and recipient port salinity combinations in the GLSLR International pathway, the GLSLR region received the greatest establishment risk reduction from exchange plus treatment when the source port was either fresh or brackish water. Exchange plus treatment performed slightly better than treatment alone when all vessels adhered to the D-2 standard. For freshwater source ports, moving from unmanaged ballast water to exchange plus treatment (100%) resulted in a 97% (zooplankton) and 89% (phytoplankton) reduction in the expected probability that at least one species establishment occurs per trip (PEPT), while treatment (100%) alone reduced PEPT 91% (zooplankton) and 79% (phytoplankton; Figures 13 and 14). For brackish source ports, relative to no-management, exchange plus treatment (100%) reduced PEPT 97% (zooplankton) and 90% (phytoplankton), whereas treatment alone lowered PEPT 90% (zooplankton) and 84% (phytoplankton). The difference in effectiveness between these two management strategies was much greater when only half of the transits were applied with the D-2 standard. For freshwater source ports, relative to unmanaged ballast water, PEPT was reduced by 82% (zooplankton) and 73% (phytoplankton) for exchange plus treatment (50%) versus reductions in PEPT of 45% (zooplankton) and 29% (phytoplankton) for treatment alone (50%). For brackish source ports, compared to no-management, PEPT decreased 79% (zooplankton) and 86% (phytoplankton) for exchange plus treatment (50%) versus reductions in PEPT of 45% (zooplankton) and 38% (phytoplankton) for treatment alone (50%).

ARCTIC INTERNATIONAL PATHWAY

Without management, 0.24 non-indigenous zooplankton and 1.98 harmful phytoplankton species were expected to become established per decade, while under BWE, establishment rates were expected to be slightly lower for zooplankton (0.21 SpPD) and similar for phytoplankton (1.91 SpPD; Figures 10 and 11). When the D-2 standard was applied to all ship-trips, invasion rates were considerably lower than BWE for both zooplankton (0.05 SpPD; 76% reduction from BWE) and phytoplankton (0.28 SpPD; 85% reduction from BWE; Figures 10-12). There was no benefit from combining exchange with treatment (100%) for harmful phytoplankton (0.23 SpPD; 88% reduction from BWE). However, treatment alone produced marginally lower risk reductions than exchange plus treatment for non-indigenous zooplankton, which had an SpPD of 0.03 (86% reduction from BWE); note that the difference in SpPD for zooplankton between these two management strategies may be negligible due to the large overlap in their confidence intervals (Figures 10 and 11). The ability of both treatment alone and exchange plus treatment to mitigate the risk of establishment was greatly reduced when only half the voyages were applied with the D-2 standard. The effect of these two scenarios on SpPD was similar to one another for harmful phytoplankton (treatment, 29% decrease from BWE vs. exchange plus treatment, 33% reduction from BWE), while exchange plus treatment was more effective at reducing SpPD of zooplankton than treatment alone (exchange plus treatment, 48% decrease from BWE vs. treatment, 14% reduction from BWE; Figure 12).

ARCTIC DOMESTIC PATHWAY

When compared to unmanaged ballast water, exchange alone increased the establishment rate of harmful phytoplankton from 0.89 (no-management) to 1.23 (exchange-only) and did not affect the invasion rate of non-indigenous zooplankton (no-management, 1.15 SpPD vs. exchange-only, 1.13 SpPD; Figures 10 and 11). Moving from BWE alone to treatment-only (50%) reduced SpPD 35% to 0.74 for zooplankton and 56% to 0.54 for phytoplankton (Figures 10-12). In this case, exchange plus treatment was less beneficial than treatment alone, which had an SpPD of 0.82 for zooplankton (27% reduction from BWE) and 0.82 for phytoplankton (33% reduction from BWE). When all vessels adhered to the D-2 standard, treatment resulted in pronounced reductions in species establishment rates for both non-indigenous zooplankton (0.04 SpPD) and harmful phytoplankton (0.15 SpPD), and the effectiveness of exchange plus treatment was equivalent to that of treatment alone (zooplankton, 0.09 SpPD; phytoplankton, 0.13 SpPD).

ALL SHIPPING PATHWAYS: SOURCE AND RECIPIENT PORT SALINITY PAIRS

Although the effectiveness of BWE greatly varied among the port salinity pairs, BWE alone often had a lower PEPT than unmanaged ballast water for zooplankton (BWE, 0.00012-0.0082 PEPT; no-management, 0.00021-0.016 PEPT) and phytoplankton (BWE, 0.00026-0.013 PEPT; no-management, 0.0006-0.01857 PEPT; Figures 15 and 16). However, BWE provided considerably less invasion risk reductions than treatment-only (100%), which had a PEPT ranging from 0-0.0021 (non-indigenous zooplankton) and from 0.000053-0.0011 (harmful phytoplankton).

When comparing the effect of BWE on invasion rates among the different source and recipient port salinity combinations examined, the trend in the results was that the effectiveness of BWE decreased as the salinity of both the source and recipient port increased. For example, considering harmful phytoplankton, the average percentage reduction in PEPT from unmanaged ballast water to BWE was 81% for freshwater-freshwater (i.e., source and recipient port pair), 70% for brackish-brackish, and 9% for marine-marine (Figure 14). Additionally, when the source or recipient port were marine, the effectiveness of BWE generally decreased. For example, moving from no-management to exchange-only increased PEPT of harmful algae 14% for marine-freshwater and 11% for brackish-marine.

The effectiveness of exchange plus treatment compared to treatment alone (when the D-2 standard was applied to either all or half of ship-trips) varied depending on the salinity of the source and recipient ports. Note that for the following paragraphs, the percentage change in PEPT for each management scenario is relative to the no-management baseline.

When all ships met the D-2 standard, exchange plus treatment was the most effective strategy at reducing non-indigenous zooplankton and harmful phytoplankton establishment rates for freshwater recipient ports when the source port was either fresh or brackish water. For freshwater-freshwater, exchange plus treatment reduced the expected per-trip establishment rate 96% to 0.00071 (zooplankton) and 98% to 0.00013 (phytoplankton), whereas treatment alone reduced PEPT 87% to 0.0021 (zooplankton) and 89% to 0.00075 (phytoplankton; Figures 13-16). For brackish-freshwater, exchange plus treatment decreased PEPT 95% for zooplankton (0.00060 PEPT) and 98% for phytoplankton (0.00012 PEPT), whereas treatment alone had slightly less reductions in PEPT of 89% for zooplankton (0.0014 PEPT) and 91% for phytoplankton (0.00044 PEPT). On the other hand, when vessels travel from marine to freshwater ports, treatment alone was the most effective strategy at reducing the per-trip establishment risk of non-indigenous zooplankton (treatment, 88% decrease in PEPT to 0.00026 vs. exchange plus treatment, 75% reduction in PEPT to 0.00054), while the addition of exchange with treatment had negligible effect on the harmful phytoplankton invasion risk per trip (treatment alone, 90% reduction in PEPT to 0.000088 vs. exchange plus treatment, 89% decrease in PEPT to 0.000097). For brackish or marine recipient ports, exchange plus treatment (100%) provided similar protection against invasions to that by the treatment-only (100%) scenario. For example, among brackish and marine recipient ports for non-indigenous zooplankton, exchange plus treatment lowered the per-trip establishment risk by 97-100%, while the risk reductions of treatment alone ranged between 97-100%. This trend was mirrored for harmful phytoplankton, with exchange plus treatment reducing PEPT by 92-98% and treatment alone lowering PEPT by 91-94%.

With the application of the D-2 standard on half of the voyages, exchange plus treatment consistently reduced the species establishment risk over treatment alone for freshwater source ports, while exchange plus treatment had varied effectiveness compared to treatment alone for brackish and marine source ports. For freshwater source ports, exchange plus treatment reduced PEPT by 74-82% to 0.000055-0.0029 (zooplankton) and 78-91% to 0.00013-0.0029 PEPT (phytoplankton), whereas treatment provided lower per-trip establishment risk reductions of 37-44% for zooplankton (0.00013-0.0089 PEPT) and 42-44% for phytoplankton (0.00035-0.0075 PEPT; Figure 13-16).

Exchange plus treatment (50%) was more effective at mitigating establishment risk than treatment alone (50%) for voyages originating from brackish ports and terminating at fresh or brackish water environments, with exchange plus treatment reducing the PEPT of non-indigenous zooplankton and harmful phytoplankton 33-39% greater than treatment alone (Figures 13 and 14). Contrarily, treatment alone performed slightly better than exchange plus treatment for brackish-marine; exchange plus treatment had a PEPT of 0.0031 for zooplankton and 0.0068 for phytoplankton, while treatment had a PEPT of 0.0028 (zooplankton) and 0.0061 (phytoplankton), resulting in a 6% greater reduction in establishment risk across both taxonomic groups for exchange plus treatment from no-management (zooplankton, 0.0054 PEPT; phytoplankton, 0.011 PEPT; Figures 13-16).

For marine source ports, treatment alone (50%) was more effective at reducing invasion risk compared to exchange plus treatment (50%) when voyages terminated at freshwater ports, while there was little difference between these two management strategies when voyages terminated at brackish or marine environments. For marine-freshwater, treatment reduced PEPT 44% (zooplankton) and 40% (phytoplankton), while lower reductions in invasion risk were observed for exchange plus treatment, which decreased PEPT 9% (zooplankton) and 28% (phytoplankton; Figures 13 and 14). Lastly, considering marine-brackish and marine-marine, across both taxonomic groups, treatment lowered PEPT 45-49%, while similar reductions in PEPT of 44-52% were produced by exchange plus treatment.

DISCUSSION

IMPLICATIONS OF BALLAST WATER MANAGEMENT ON ECOLOGICAL OUTCOMES

As observed with many ecological management strategies, the effectiveness of the processes for managing ballast water in this study were not uniform across regions, habitat types, or taxonomic groups. The D-2 standard provided superior and more consistent protection against the invasion risk of harmful species compared to BWE. In certain circumstances, exchange plus treatment provided the greatest reductions in invasion risk in GLSLR International and Arctic International, while the incorporation of exchange with treatment did not provide additional benefit in the Pacific International, Atlantic International, and Arctic Domestic pathways. Exchange plus treatment, with the application of the D-2 standard on all transits, was the most effective strategy at mitigating invasions when both the source and recipient environments were of low salinity (i.e., freshwater-freshwater or brackish-freshwater), while this strategy did not provide enhanced protection when high salinity ports were involved. In the event that only 50% of transits adhered to the D-2 standard, exchange plus treatment provided superior invasion risk reductions than treatment alone when the source ports were fresh or brackish water.

In the event that all vessels adhered to the D-2 standard, the number of decadal species establishments of non-indigenous zooplankton and harmful phytoplankton was substantially reduced from that of BWE for each shipping pathway. Furthermore, treatment typically provided greater and more consistent protection against harmful species invasions among different habitat types. The effectiveness of BWE and the D-2 standard for various shipping pathways in Canada were previously modelled by Casas-Monroy et al. (2014). The results of this study were similar to that of Casas-Monroy et al. (2014) where they determined that the establishment risk of non-indigenous zooplankton was greatly reduced for each shipping pathway examined under the D-2 standard compared to BWE. Unlike in this study, Casas-Monroy et al. (2014) determined that the invasion risk of phytoplankton was not reduced for voyages arriving to the Pacific or Atlantic Coasts under the D-2 standard compared to BWE alone, since the mean abundance of phytoplankton in ballast water was already within the limit of the D-2 standard. The difference between these two results largely stems from the breadth of the analyses, with the earlier study examining only non-indigenous dinoflagellates whilst this study included harmful diatoms and dinoflagellates.

In the GLSLR International pathway, utilizing exchange plus treatment provided slightly greater protection against the establishment of non-indigenous zooplankton than treatment alone when all ships adhered to the D-2 standard, while this multidimensional strategy provided the greatest protection against both harmful phytoplankton and non-indigenous zooplankton invasions when only half the ship-trips were applied with the D-2 standard. The observed benefit of exchange plus treatment likely occurred because over 40% of the voyages were comprised of freshwater-freshwater and brackish-freshwater, where exchange plus treatment was more effective than treatment alone due to the effect of BWE on establishment rates. The overall effect of exchange in this pathway was not surprising, as the rate of species invasions attributed to discharged ballast water has markedly decreased in the GLSLR since BWE (and flushing of residual ballast) has been implemented (Bailey et al. 2011). BWE is highly effective at mitigating establishment risk when both the source and recipient ports are of low salinity, as exchange will create a salinity barrier by introducing highly saline marine waters, which can, a) reduce the survival of low salinity organisms in tanks, should they continue to exist beyond the flush period, or b) introduce new viable marine organisms that are unlikely to survive after release into freshwater ecosystems (Santagata et al. 2008, Ellis and MacIsaac 2009, Bailey et al. 2011, Reid 2012).

For the Arctic International pathway, exchange plus treatment lowered the expected number of decadal non-indigenous zooplankton invasions greater than did treatment alone, when all or half the transits were applied with the D-2 standard; exchange plus treatment did not provide substantial benefits over treatment alone for harmful phytoplankton in this pathway. The model results of this pathway have greater uncertainty relative to the Atlantic, Pacific, or GLSLR pathways due to the limited biological data for this region.

Exchange plus treatment produced either similar or fewer reductions in the number of harmful species establishments per decade compared to treatment alone in the Pacific International, Atlantic International, or Arctic Domestic pathways, when the D-2 standard was applied to all or half of the transits. This was likely due to BWE having little effect on invasion risk in these regions because the vast majority of their transits were terminated at marine ports, where BWE was less effective relative to freshwater ports. Since our model only considered the environmental mismatching effect of mid-ocean exchange, its effectiveness was typically greatly reduced when either (or both) the source or/and recipient port was of high salinity. This corresponds with studies that concluded BWE is less effective at protecting high saline ecosystems, as post-exchange ballast water may contain a high abundance of non-indigenous or harmful oceanic or residual coastal species that have a high risk of establishing in brackish or marine recipient ecosystems (Cordell et al. 2009, Simard et al. 2011, Reid 2012, Roy et al. 2012, Casas-Monroy et al. 2016).

Since environmental mismatch through differences in salinity is one of the primary mechanisms of action of BWE, the effectiveness of exchange plus treatment compared to treatment alone at mitigating the risk of invasions was context-specific.

Freshwater-freshwater generally received the greatest invasion risk reduction from exchange plus treatment for both taxonomic groups examined, when all vessels met the D-2 standard. The enhanced efficacy of exchange plus treatment for freshwater-freshwater was supported by land-based and ship-board studies conducted by Briski et al. (2013 and 2015, respectively), who determined that ballast tanks contained mainly lower risk marine taxa after performing exchange plus treatment, and mainly higher risk freshwater and euryhaline taxa after performing treatment (risk relative to freshwater recipient environments). Here, we report a new finding that exchange plus treatment may be more beneficial than treatment alone for brackish-freshwater, albeit to a slightly lesser extent than freshwater-freshwater. On the other hand, exchange plus treatment had varied effectiveness compared to treatment alone for all other port salinity pairs when all shipping pathways were combined, although there were only minor differences in establishment risk between these two management strategies, except for marine-freshwater, which received the greatest protection against non-indigenous zooplankton invasions from treatment alone.

In the event that only 50% of transits were applied with the D-2 standard, exchange plus treatment was the most effective strategy at reducing harmful species establishments when the ballast source was either fresh (regardless of the salinity of the recipient port) or brackish water (for fresh or brackish recipient ports only) when all shipping pathways were combined. The effectiveness of exchange plus treatment was less prominent for brackish source ports, as it resulted in a slightly higher PEPT than treatment alone for brackish-marine. Interestingly, the results for freshwater-marine indicated a benefit of exchange when the salinity mismatch was expected to be greatly reduced, resulting in greater PEPT reductions for exchange plus treatment than treatment alone. It is possible that this reduction in species establishment is due to a temperature mismatch, wherein the temperature of the source water more closely matches the destination port, than it does the temperature in the exchange location. On the other hand, for marine source ports, treatment was more effective than exchange plus treatment when voyages terminated at freshwater environments, while both these management strategies had similar effectiveness when vessels arrived to brackish or marine ports.

GROUND-TRUTHING

It is important to ground-truth any model. Unfortunately, this is a difficult task when modelling species invasions because of the lack of data concerning failed invasions and the many problems with observational species discovery data (inconsistent search effort, detection bias, etc.). In other words, understanding the reality of the conditions of species establishment is difficult, and hence the mechanistic approach was used in this document to estimate invasion based on biological processes known to be relevant. Nonetheless, comparing the model with observational species discovery data in the Great Lakes provided an opportunity to understand correspondence between the model results and a representation of the number of invasions through time.

Ricciardi (2006) used observed species discovery data in the Great Lakes region to estimate the number of ship-mediated non-indigenous species (vascular aquatic plants, algae, invertebrates, and fishes) in the Great Lakes basin for the years 1840 to 2003. For the period around 1960, Ricciardi (2006) estimated an invasion rate of 1.0 sp/yr and, from 1993 to 2004, an estimated 1.2 sp/yr (0.9 sp/yr for ‘free living species’). The results of the mechanistic model presented here (Figures 10 and 11) estimate a rate of 1.09 sp/yr during the pre-2006 condition (no-management), and a rate of 0.94 sp/yr during the post-2006 condition (exchange-only); these values were determined by summing the zooplankton and phytoplankton expected invasion rates, then, using Cohen and Carlton’s (1995) 40% and 20% method for estimating the number of non-indigenous species of aquatic plants (40% of total plankton rates) and fish (20% of total plankton rates), respectively. As species discovery data will always underestimate the number of true established species, the baseline no-management model of 1.09 is reflective of the Ricciardi (2006) result. The hypotheses for the deviation of the results of this model from Ricciardi's post-exchange results (0.94 sp/yr vs. 1.2 sp/yr) are: 1) a time-lag associated with species discovery led to missed detection of some invading species via field sampling programs, which is a demonstrated problem with cryptic species (i.e., some species detected after exchange regulations may have invaded prior to exchange regulations); 2) substantial deviations in past shipping histories led to historical conditions that deviated from the model; or 3) the estimated invasion rate from 1993 to 2004 in Ricciardi (2006) was prior to the implementation of ballast tank flushing regulations in 2006, whereas the model for this current study does not consider the ecological impacts of ships with no declarable ballast on board. Nevertheless, the close correspondence between the model results and Ricciardi (2006) indicates that the model provides a sound representation of the factors leading to ballast-mediated invasions in Canada.

MODEL SENSITIVITY

Sensitivity analyses were conducted on parameters that could change the number of species establishments, but not the relative effectiveness of the management scenarios (part 1), and parameters that could change the relative effectiveness of the management scenarios and thus alter management decisions (part 2).

For part 1, parameters included altering the sample density in ballast tanks, proportion of species in tanks that are harmful, volume of shipping traffic, patterns of shipping traffic, per-propagule likelihood of establishment (α) and the shape parameter for Allee effect (c). These parameters were each shown to separately have a low effect on model outcome (most showed < 10% change in expected establishments for a 25% parameter shift), except changes in α and c (Table 6). When all species were assigned an α equal to 0.05, which may be expected if all species reproduced clonally, establishments increased by 22-fold on average. We do not expect that this is a realistic assumption; regardless, it would not change the relative performance of the management options. Likewise, changing *c* to a value of 2, increased establishments 86% on average, but did not change the relative performance of treatment methods. Note that in some cases, a larger percentage change was observed for certain treatment types in a given sensitivity analysis (e.g., for 25% decrease in transits, there were ~8% reductions for no-management, exchange-only, treatment-only [50%], and exchange plus treatment [50%], and ~25% reductions for treatment-only [100%] and exchange plus treatment [100%]). These results occurred where there was a similar magnitude reduction across all management scenarios, but since the scenarios involving treatment (100%) had fewer initial establishments, this resulted in a greater percentage change in SpPD.

It is worthwhile to note that while our sensitivity analysis examined individual changes separately, it is possible that multiple simultaneous changes from our assumed model parameters could be necessary, due to either incorrectly assumed parameters or changes in real world conditions. Multiple changes could lead to an additive or multiplicative effect on expected outcomes or, equally, could negate each other (e.g. an increase in population density with a decrease in shipping traffic may not change the overall number of establishments).

For the second part of the sensitivity analysis, the effectiveness of BWMS was altered in the treatment-only (50%) scenario by utilizing post-treatment data for ships that ‘pass’ or ‘fail’ the D-2 standard. The expected reductions in non-indigenous zooplankton establishments under this scenario reduced SpPD 44% from the baseline no-management, which lies between the expected risk reductions for the standard treatment-only scenarios that lowered SpPD 21% (D-2 standard applied to half of the transits) and 85% (all ships met the D-2 standard). As the sensitivity analysis was based on preliminary data from trials in the early stages of utilizing BWMS, its results should be considered with caution. However, this demonstrates how the expected reduction in establishments is affected by the efficacy of BWMS and how the number of species establishments may be expected to change as BWMS become more reliable.

SOURCES OF UNCERTAINTY

It is recognized that there is uncertainty associated with the probability distribution used to describe the per-capita establishment probability (α) across harmful species in a ballast tank, as different groups of species may not be uniformly distributed on the curve. Given data limitations, it was not possible to create specific α probability distributions for each group of species. For instance, the true mean α value for phytoplankton may have been underestimated since most of these species reproduce asexually. This uncertainty was addressed in the sensitivity analysis by setting α to 0.05 for all species, which is equivalent to the estimated α for parthenogenetic species. Additionally, groups of species inhabiting different habitats may have different α probability distributions, increasing the uncertainty in the results. For example, there are more parthenogenetic species in freshwater than in marine ecosystems, resulting in a higher mean α value for freshwater species compared to marine species. The uncertainty associated with the α probability distribution affects the species establishment rates and relative importance of the zooplankton/phytoplankton pathways. It could potentially also alter the relative effectiveness of the management scenarios, if variability is expected between mid-ocean taxa and species from freshwater or coastal source ports since it could then alter the outcome after BWE. Therefore, having more data to estimate distinct α distributions for the taxonomic groups under consideration would improve the overall establishment estimates and comparisons of the effectiveness of the management strategies (see Considerations for Re-Running the Model section for more details).

Greater uncertainty is associated with the Arctic results compared to other Canadian regions due to limited or unavailable biological data Also, Arctic shipping activity is expected to increase in the future, which may disproportionately increase the number of discharge events and diversity of source ports (i.e., changing the composition of ballast water species transported to the Arctic) relative to other geographical regions in Canada.

Other sources of uncertainty include the high variability in organism abundance among ship-trips. It is recognized that plankton population sizes fluctuate seasonally, influencing invasion risk (Zhang and Dickman 1999, Simard et al. 2011), but this factor is beyond the scope of this study. There is also uncertainty surrounding the environmental data used in this model, as environmental conditions (i.e., salinity and temperature) vary spatially and temporally, and are rarely port-specific. Lastly, this model estimated species survival based on the environmental match in temperature and salinity, however other biotic and abiotic variables that influence species survival and establishment – including more refined measures of habitat suitability, such as nutrient availability, competition, or predation – were not considered, contributing to the uncertainty in the model results. This limitation is relevant for interpreting the effect of BWE, where the potential for mid-ocean species to establish in recipient ports was based solely on salinity and thermal matching and did not consider how other factors necessary for survival or establishment would deviate between nearshore and mid-ocean ecosystems.

CONSIDERATIONS FOR RE-RUNNING THE MODEL

While the model is relatively robust to changes in its input parameters, prominent real-world changes for any of the parameters will lead to different outputs from the forecast presented in this study and, in such cases, would necessitate re-running this model to update the results accordingly. For example, major changes in shipping traffic patterns, where: 1) large increases of ship traffic (e.g., doubling or tripling in the number of transits) would increase the expected number of annual species invasions; or 2) changes to the composition of port salinity combinations within a given shipping pathway may change the relative effectiveness of management strategies, given the influence of environmental salinity on the effectiveness of BWE (e.g., the effectiveness of exchange plus treatment would be expected to increase in the Pacific International pathway if there is an increase in freshwater-freshwater transits). However, the likely consequence of changes to the composition of port salinity combinations can be inferred from the salinity-specific results in Figures 15 and 16. Additionally, if the frequency with which ships meet the D-2 standard or the organism concentration of ballast water deviates from their respective values used in this study, the results could be updated to reflect the changes in invasion risk.

Another example of an input parameter alteration that would warrant the results to be revisited would be the observation of a higher proportion of parthenogenetic species within the establishment distribution. To address this issue, the National Research Council (2011) recommended that a series of field experiments be undertaken to determine the establishment values across a range of species and environmental conditions. These studies would help to inform the shape of the beta distribution used in the establishment section of the model. Should a high proportion of low values exist (signifying extinction for most species at small initial population size), it is likely that the current beta parameters and therefore model output would not change. However, should a high proportion of high values exist, signifying the ability of many transported species to establish new populations at low densities, a new beta distribution to reflect the increased establishment risk should be incorporated. Therefore, as new scientific information becomes available, the model parameters should be re-evaluated to ensure that the model remains reflective of the observed behaviour of the system. Until new parameter values become available, the sensitivity analysis (Table 6) provides guidance on how the model outcomes will change due to shifts in parameter values.

This model assumed that the only effect of BWE was a change in survival probability reflective of the environmental correspondence between the locations of exchange and recipient ports. However, incorporating the full effect of BWE, such as its effect on population densities, may more accurately contrast the establishment rates of mid-ocean species to coastal or freshwater species, influencing the relative performance of the management strategies. The next possible update for this research is to incorporate a comprehensive modelled effect of BWE, which will allow more accurate comparisons of the effectiveness of the management methods.

Results of this study may also be updated if new input data becomes accessible. For example, it is recommended that these results be updated when improved biological data is available, as there was limited biological data for the Arctic pathways. Although this report used the best available port environmental data, if more refined environmental data becomes available, or environmental conditions change, these factors could influence establishment outcomes. Thus, if new environmental data that differs from the inputs used in this study becomes available, this model should be re-run to update the results.

BALLAST WATER EXCHANGE PLUS TREATMENT PROTOCOLS

As outlined in the introduction, the two protocols for ships to undertake exchange plus treatment are E+T and T+E+T. Given the large relative differences between the management scenarios involving the application of the D-2 standard on 100% and 50% of ship-trips within each shipping pathway, it is important to maintain the highest functionality of BWMS. Performing T+E+T may increase the frequency of BWMS malfunctions during challenging port water conditions (e.g., due to high turbidity), whereas conducting E+T may reduce BWMS malfunctions, since mid-ocean water is generally less difficult to treat than port water (C. Wiley, past Chair of IMO Ballast Water Review and Working Groups, pers. comm., Briski et al. 2013). Therefore, E+T is the recommended protocol to be conducted by vessels. It is recognized that this protocol conflicts with the current regulations of the IMO Convention.

CONCLUSION

The output variables used in this model (the per-trip probability of establishment and the number of species establishments per decade) are relevant metrics to document the establishment of invasive species. As demonstrated throughout this report, these output variables are strongly influenced by ballast water and its management via exchange, treatment, or the combination of both methods. Moreover, because a mechanistic approach was undertaken, changes in expected values can definitively be attributed to each ballast water management strategy, given the assumptions, parameters, and model structure. While factors beyond those considered in the model may influence invasion risk, this assessment incorporates a core set of factors (the hierarchy of species arrival, survival, and establishment) that influence invasions while integrating relevant management mechanisms within this hierarchy (species survival: exchange, and species establishment via initial population size: treatment). Given the prominent effect of ballast water management observed in this study, the scenarios provide logical starting points to evaluate the effect of different management strategies to reduce the risk of ballast-mediated invasions in Canadian aquatic ecosystems. The choice of specific management strategy is a risk tolerance decision related to the acceptable degree of human-mediated change on ecological processes, the value associated with ecosystem services, and additional social and economic factors.

RECOMMENDATIONS

To support current and future research and science advice, it is important to collect more comprehensive data on the transport of ballast water within and to Canada (e.g., ballast source and volume). Importantly, if some ships are unable to adhere to the D-2 standard, it is recommended that biological data from treated ballast water be collected for a sample of vessels to further inform species invasion risks related to exchange plus treatment.

REFERENCES CITED

Adams, J.K., Ellis, S.M., Chan, F.T., Bronnenhuber, A.G., Doolittle, J.E., Simard, N., McKenzie, C.H, Martin, J.L., and Bailey, S.A. 2014. Relative risk assessment for ship-mediated introductions of aquatic nonindigenous species to the Atlantic Region of Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/116. v + 403 p.

Adebayo, A.A., Zhan, A., Bailey, S.A., and MacIsaac, H.J. 2014. Domestic ships as a potential pathway of nonindigenous species from the Saint Lawrence River to the Great Lakes. Biol. Invasions 16(4): 793–801. doi:10.1007/s10530-013-0537-5

Arctic Council. 2009. Arctic marine shipping assessment 2009 report. Arctic Council, Tromsø, Norway. 194 p.

Bailey, S.A. 2015. An overview of thirty years of research on ballast water as a vector for aquatic invasive species to freshwater and marine environments. Aquat. Ecosyst. Health Manag. 18(3): 261–268. doi:10.1080/14634988.2015.1027129

Bailey, S.A., Chan, F., Ellis, S.M., Bronnenhuber, J.E., Bradie, J.N. and Simard, N. 2012. Risk assessment for ship-mediated introductions of aquatic nonindigenous species to the Great Lakes and freshwater St. Lawrence River. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/104. vi + 224 p.

Bailey, S.A., Deneau, M.G., Jean, L., Wiley, C.J., Leung, B., and Macisaac, H.J. 2011. Evaluating efficacy of an environmental policy to prevent biological invasions. Environ. Sci. Technol. 45(7): 2554–2561. doi:10.1021/es102655j

Bailey, S.A., Vélez-Espino, L.A., Johannsson, O.E., Koops, M.A., and Wiley, C.J. 2009. Estimating establishment probabilities of Cladocera introduced at low density: An evaluation of the proposed ballast water discharge standards. Can. J. Fish. Aquat. Sci. 66(2): 261–276. doi:10.1139/f08-200

Bolker, B.M. 2008. Ecological models and data in R. Princeton University Press, Princeton, NJ. 408 p.

Bradie, J., Chivers, C., and Leung, B. 2013. Importing risk: Quantifying the propagule pressure-establishment relationship at the pathway level. Diversity Distrib. 19(8): 1020–1030. doi:10.1111/ddi.12081

Bradie, J., Pietrobon, A., and Leung, B. 2015. Beyond species-specific assessments: an analysis and validation of environmental distance metrics for non-indigenous species risk assessment. Biol. Invasions 17(12): 3455–3465.

Briski, E., Allinger, L.E., Balcer, M., Cangelosi, A., Fanberg, L., Markee, T.P., Mays, N., Polkinghorne, C.N., Prihoda, K.R., Reavie, E.D., Regan, D.H., Reid, D.H., Saillard, H.J., Schwerdt, T., Schaefer, H., TenEyck, M., Wiley, C.J., and Bailey, S.A. 2013. Multidimensional approach to invasive species prevention. Environ. Sci. Technol. 47: 1216–1221. doi:10.1021/es3029445

Briski, E., Bailey, S.A., Casas-Monroy, O., DiBacco, C., Kaczmarska, I., Levings, C., MacGillivary, M.L., McKindsey, C.W., Nasmith, L.E., Parenteau, M., Piercey, G., Rochon, A., Roy, S., Simard, N., Villac, M.C., Weise, A., and MacIsaac, H.J. 2012a. Relationship between propagule pressure and colonization pressure in invasion ecology: A test with ships’ ballast. Proc. R. Soc. B 279: 2990–2997. doi:10.1098/rspb.2011.2671

Briski, E., Gollasch, S., David, M., Linley, R.D., Casas-Monroy, O., Rajakaruna, H., and Bailey, S.A. 2015. Combining ballast water exchange and treatment to maximize prevention of species introductions to freshwater ecosystems. Environ. Sci. Technol. 49: 9566–9573. doi:10.1021/acs.est.5b01795

Briski, E., Wiley, C.J., and Bailey, S.A. 2012b. Role of domestic shipping in the introduction or secondary spread of nonindigenous species: Biological invasions within the Laurentian Great Lakes. J. Appl. Ecol. 49(5): 1124–1130. doi:10.1111/j.1365-2664.2012.02186.x

Cangelosi, A., Allinger, L., Balcer, M., Mays, N., Markee, T., Polkinghorne, C., Prihoda, K., Reavie, E., Reid, D., Saillard, H., Schwerdt, T., Schaefer, H., and TenEyck, M. 2011. Final report of the land-based, freshwater testing of the AlfaWall AB PureBallast® ballast water treatment system. Great Ships Initiative, Northeast-Midwest Institute, Washington, DC. 94 p.

Cangelosi, A., Anders, O., Balcer, M., Beesley, K., Fanberg, L., Gebhard, S., and Venditto, T. 2018. Great Lakes ship ballast monitoring project, Technical Report. LSRI/GWRC/TR/GLSBM/1, Lake Superior Research Institute, University of Wisconsin, USA.88 p.

Casas-Monroy, O. 2012. Introduction des dinoflagellés non-indigènes dans les écosystèmes aquatiques canadiens via les réservoirs de ballasts de navires. Thesis (Ph.D.) Université du Québec à Rimouski, Rimouski, Que. 150 p.

Casas-Monroy, O., Linley, R.D., Adams, J.K., Chan, F.T., Drake, D.A.R., and Bailey, S.A. 2014. National risk assessment for introduction of aquatic nonindigenous species to Canada by ballast water. DFO Can. Sci. Advis. Sec. Res. Doc. 2013/128. vi + 73 p.

Casas-Monroy, O., Linley, R.D., Chan, P., Kydd, J., Byllaardt, J.V., and Bailey, S. 2018. Evaluating efficacy of filtration UV-C radiation for ballast water treatment at different temperatures. J. Sea Res. 133: 20–28. doi:10.1016/j.seares.2017.02.001

Casas-Monroy, O., Parenteau, M., Drake, D.A.R., Roy, S., and Rochon, A. 2016. Absolute estimates of the propagule pressure of viable dinoflagellates across Canadian coasts: The variable influence of ballast water exchange. Mar. Biol. 163(8): 1-15. doi:10.1007/s00227-016-2946-3

Chan, F.T., Bronnenhuber, J.E., Bradie, J.N., Howland, K., Simard, N. and Bailey, S.A. 2012. Risk assessment for ship-mediated introductions of aquatic nonindigenous species to the Canadian Arctic. DFO Can. Sci. Advis. Sec. Res. Doc. 2011/105. pp vi + 93.

Chan, F.T., Macisaac, H.J., and Bailey, S.A. 2015. Relative importance of vessel hull fouling and ballast water as transport vectors of nonindigenous species to the Canadian Arctic. Can. J. Fish. Aquat. Sci. 72(8): 1230–1242. doi:10.1139/cjfas-2014-0473

Cohen, A.N., and Carlton, J.T. 1995. Nonindigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta – Biological Study. Report No. PB96-166525, Final Report to the U.S. Department of the Interior, U.S. Fish and Wildlife Service, Washington, D.C. 25 p.

Cordell, J.R., Lawrence, D.J., Ferm, N.C., Tear, L.M., Smith, S.S., and Herwig, R.P. 2009. Factors influencing densities of non-indigenous species in the ballast water of ships arriving at ports in Puget Sound, Washington, United States. Aquatic Conserv: Mar. Freshw. Ecosyst. 19(3): 322–343. doi:10.1002/aqc.986

de Lafontaine, Y., Despatie, S., Veilleux, É, and Wiley, C. 2008. Onboard ship evaluation of the effectiveness and the potential environmental effects of PERACLEAN® Ocean for ballast water treatment in very cold conditions. Environmental Toxicology 24: 49–65. doi:10.1002/tox.20394

Dextrase, A.J., and Mandrak, N.E. 2006. Impacts of alien invasive species on freshwater fauna at risk in Canada. Biological Invasions 8(1): 13–24. doi:10.1007/s10530-005-0232-2

DiBacco, C., Humphrey, D.B., Nasmith, L.E., and Levings, C.D. 2012. Ballast water transport of non-indigenous zooplankton to Canadian ports. ICES Journal of Marine Science 69: 483–491. doi:10.1093/icesjms/fsr133

Drake, D.A.R., Bailey, S.A., and Mandrak, N.E. 2017. Ecological Risk Assessment of Recreational Boating as a Pathway for the Secondary Spread of Aquatic Invasive Species in the Great Lakes Basin. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/030. v + 85 p.

Drake, D.A.R., Casas-Monroy, O., Koops, M.A., and Bailey, S.A. 2015. Propagule pressure in the presence of uncertainty: extending the utility of proxy variables with hierarchical models. Methods Ecol. Evol. 6(11): 1363–1371.

Drake, D.A.R., Chan, F.T., Briski, E., Bailey, S.A., and MacIsaac, H.J. 2014. Assemblage structure: An overlooked component of human-mediated species movements among freshwater ecosystems. J. Limnol. 73(s1): 108–115. doi:https://doi.org/10.4081/jlimnol.2014.802

Drake, D.A.R., and Mandrak, N.E. 2014. Bycatch, bait, anglers, and roads: quantifying vector activity and propagule introduction risk across lake ecosystems. Ecol. Appl. 8(4): 877–94.

Duggan, I., Bailey, S., Overdijk, C.V., and Macisaac, H. 2006. Invasion risk of active and diapausing invertebrates from residual ballast in ships entering Chesapeake Bay. Mar. Ecol. Prog. Ser. 324: 57–66. doi:10.3354/meps324057

Duggan, I.C., Overdijk, C.D., Bailey, S.A., Jenkins, P.T., Limén, H., and Macisaac, H.J. 2005. Invertebrates associated with residual ballast water and sediments of cargo-carrying ships entering the Great Lakes. Can. J. Fish. Aquat. Sci. 62(11): 2463–2474. doi:10.1139/f05-160

Ellis, S., and Macisaac, H.J. 2009. Salinity tolerance of Great Lakes invaders. Freshwater Biology 54: 77–89. doi:10.1111/j.1365-2427.2008.02098.x

GISIN (Global Invasive Species Information Network). 2014. Global invasive species information network database [online]. Available from http://www.gisin.org (accessed October 22, 2014).

Gray, D.K., Johengen, T.H., Reid, D.F., and MacIsaac, H.J. 2007. Efficacy of open-ocean ballast water exchange as a means of preventing invertebrate invasions between freshwater ports. Aquat. Invasions 52(6): 2386–2397.

Gregg, M., Rigby, R., and Hallegraeff, G. 2009. Review of two decades of progress in the development of management options for reducing or eradicating phytoplankton, zooplankton and bacteria in ships ballast water. Aquatic Invasions 4(3): 521–565. doi:10.3391/ai.2009.4.3.14

Humphrey, D.B. 2008. Characterizing ballast water as a vector for nonindigenous zooplankton transport. Thesis (M.Sc.)The University of British Columbia, Vancouver, B.C. 103 p.

IMO (International Maritime Organization). 2004. International convention for the control and management of ships' ballast water and sediments, 2004. International Conference on Ballast Water Management for Ships, BWM/CONF/36, London, UK. 36 p.

IMO (International Maritime Organization). 2010. Development of guidelines and other documents for uniform implementation of the 2004 BWM convention. Proposal to utilize ballast water exchange in combination with a ballast water management system to achieve an enhanced level of protection. Submitted by Canada. Sub-committee on bulk liquids and gases, 15th session, BLG 15/5/7.

IMO (International Maritime Organization). 2018. International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWM). Available from http://www.imo.org/en/About/Conventions/ListOfConventions/Pages/International-Convention-for-the-Control-and-Management-of-Ships'-Ballast-Water-and-Sediments-(BWM).aspx (accessed April 2, 2018).

IMO (International Maritime Organization). 2019. BWM Technologies, Table 3: List of ballast water management systems which received Type Approval Certification by their respective Administrations. Available from http://www.imo.org/en/OurWork/Environment/BallastWaterManagement/Documents/Table%20of%20BA%20FA%20TA%20updated%20January%202019.pdf (accessed July 4, 2019).

Jerde, C., Bampfylde, C., and Lewis, M. 2009. Chance establishment for sexual, semelparous species: Overcoming the Allee effect. The American Naturalist 173(6): 734–746. doi:10.1086/598496

Keller, R.P., Drake, J.M., Drew, M.B., and Lodge, D.M. 2011. Linking environmental conditions and ship movements to estimate invasive species transport across the global shipping network. Diversity Distrib. 17: 93–102. doi:10.1111/j.1472-4642.2010.00696.x

Klein, G., Kaczmarska, I., and Ehrman, J.M. 2009. The diatom Chatoceros in ships' ballast waters-survivorship of stowaways. Acta Botanica Croatia 68(2): 325–338.

Lee, H. II, Reusser, D.A., and Frazier, M. 2013. Approaches to setting organism-based ballast water discharge standards. Ecol. Appl. 23: 301–310.

Leung, B., Drake, J.M., and Lodge, D.M. 2004. Predicting invasions: Propagule pressure and the gravity of allee effects. Ecology 85: 1651–1660. doi:10.1890/02-0571

Linley, R.D., Doolittle, A.G., Chan, F.T., O'Neill, J., Sutherland, T., and Bailey, S.A. 2014. Relative risk assessment for ship-mediated introductions of aquatic nonindigenous species to the Pacific Region of Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2013/043. v + 208 p.

Locarnini, R.A., Mishonov, A.V., Antonov, J.I., Boyer, T.P., Garcia, H.E., Baranova, O.K., Zweng, M.M., Paver, C.R., Reagan, J.R., Johnson, D.R., Hamilton, M., and Seidov, D. 2013. World Ocean Atlas 2013, Volume 1: Temperature. *In* World Ocean Atlas 2013. *Edited by* S. Levitus and A. Mishonov. NOAA Atlas NESDIS 73. 40 p.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M., and Bazzaz, F.A. 2000. Biotic invasions: Causes, epidemiology, global consequences, and control. Ecol. Appl. 10(3): 689–710. doi:10.2307/2641039

Marson, D., Cudmore, B., Drake, D.A.R., and Mandrak, N.E. 2009. Summary of a survey of water garden owners in Canada. Can. Manuscr. Rep. Fish. Aquat. Sci. 2906: v + 23 p.

McCollin, T., Shanks, A.M., and Dunn, J. 2008. Changes in zooplankton abundance and diversity after ballast water exchange in regional seas. Marine Pollution Bulletin 56: 834–844. doi:10.1016/j.marpolbul.2008.02.004

Mills, E.L., Leach, J.H., Carlton, J.T., and Secor, C.L. 1993. Exotic species in the Great Lakes: A history of biotic crises and anthropogenic introductions. J. Great Lakes Res 19(1): 1–54. doi:10.1016/s0380-1330(93)71197-1

Mouawad Counsulting, 2013. Assessment of ballast water treatment processes and availability with respect to the Great Lakes and St. Lawrence Seaway System. *In* Transactions on ballast water treatment systems for the Great Lakes-St. Lawrence Seaway System. Report No. 2013-2-1-2, Revision No. 5, submitted to Transport Canada, Ottawa, ON. pp 21-86.

National Ballast Information Clearinghouse (2002) National ballast information clearinghouse database [online]. Available from https://nbic.si.edu/database/ (accessed 15 June 2013).

NRC (National Research Council). 1996. Stemming the tide: Controlling introductions of nonindigenous species by ships' ballast water. National Academy Press, Washington, D.C. xvi + 141 p.

NRC (National Research Council). 2011. Assessing the relationship between propagule pressure and invasion risk in ballast water. *Edited by* Carlton, J.T., Ruiz, G.M., Byers, J.E., Cangelosi, A., Dobbs, F.C., Grosholz E.D., Leung, B., MacIsaac, H.J., & Wonham, M.J., eds. National Research Council (USA), Water Science and Technology Board, U.S. National Academies. 123 p.

PAME (Protection of the Arctic Marine Environment Working Group). 2013. Large marine ecosystems (LMEs) of the Arctic area: Revision of the Arctic LME map (2nd ed.). Arctic Council, Akureyri, Iceland. 19 p.

Paolucci, E.M., Hernandez, M.R., Potapov, A., Lewis, M.A., and Macisaac, H.J. 2015. Hybrid system increases efficiency of ballast water treatment. J. Appl. Ecol. 52(2): 348–357. doi:10.1111/1365-2664.12397

Paolucci, E.M., Ron, L., and Macisaac, H.J. 2017. Combining ballast water treatment and ballast water exchange: Reducing colonization pressure and propagule pressure of phytoplankton organisms. Aquat. Ecosyst. Health Manag.20(4): 369-377. doi:10.1080/14634988.2017.1404419

Reid, D.F. 2012. The role of osmotic stress (salinity shock) in protecting the Great Lakes from ballast-associated aquatic invaders. Submitted to the U.S. Saint Lawrence Seaway Development Corporation, U.S. Department of Transportation. iv + 31 p.

Remane, A., and Schlieper, C. 1972. Biology of brackish water (second edition). Schweizerbart, Stuttgart, Germany. 372 p.

Ricciardi, A. 2006. Patterns of invasion in the Laurentian Great Lakes in relation to changes in vector activity. Diversity Distrib. 12(4): 425–433. doi:10.1111/j.1366-9516.2006.00262.x

Roy, S., Parenteau, M., Casas-Monroy, O., and Rochon, A. 2012. Coastal ship traffic: A significant introduction vector for potentially harmful dinoflagellates in eastern Canada. Can. J. Fish. Aquat. Sci. 69(4): 627–644. doi:10.1139/f2012-008

Ruiz, G.M., and Smith, G. 2005. Biological study of container ships arriving to the Port of Oakland. Part B – ballast water exchange efficacy: Results of tests on eight container ships. A pilot study. Smithsonian Environmental Research Center, Edgewater, Maryland.

Ruiz, G.M., Smith, G.E., Verling, E., and Santagata, S. 2007. Efficacy of ballast water exchange. *In* Current state of understanding about the effectiveness of ballast water exchange (BWE) in reducing aquatic nonindigenous species (ANS) introductions to the Great Lakes Basin and Chesapeake Bay, USA: Synthesis and analysis of existing information. *Edited b*y G.M. Ruiz and D.F. Reid. NOAA Technical Memorandum GLERL-142. NOAA GLERL, Ann Arbor, Mich. pp 27–43.

Santagata, S., Gasiūnaite, Z., Verling, E., Cordell, J., Eason, K., Cohen, J., Bacela, K., Quilez-Badia, G., Johengen, T.H., Reid, D.F., and Ruiz, G. 2008. Effect of osmotic shock as a management strategy to reduce transfers of non-indigenous species among low-salinity ports by ships. Aquatic Invasions 3(1): 61–76. doi:10.3391/ai.2008.3.1.10

Simard, N., Plourde, S., Gilbert, M., and Gollasch, S. 2011. Net efficacy of open ocean ballast water exchange on plankton communities. Journal of Plankton Research 33(9): 1378–1395. doi:10.1093/plankt/fbr038

Transport Canada. 2007. A Guide to Canada’s Ballast Water Control and Management Regulations. Edition 7, TP 13617E: iii + 46 p.

USEPA (United States Environmental Protection Agency). 2017a. *Brachionus leydigii*. https://www.epa.gov/great-lakes-monitoring/brachionus-leydigii (accessed 30 December 2017).

USEPA (United States Environmental Protection Agency). 2017b. *Thermocyclops crassus*. https://www.epa.gov/great-lakes-monitoring/brachionus-leydigii (accessed 30 December 2017).

Wonham, M.J., Byers, J.E., Grosholz, E.D., and Leung, B. 2013. Modeling the relationship between propagule pressure and invasion risk to inform policy and management. Ecol. Appl. 23: 1691–1706. doi:10.1890/12-1985.1

Wonham, M.J., Walton, W.C., Ruiz, G.M., Frese, A.M., and Galil, B.S., 2001. Going to the source: role of the invasion pathway in determining potential invaders. Mar. Ecol. Prog. Ser. 215: 1–12.

Zhang, F., and Dickman, M. 1999. Mid-ocean exchange of container vessel ballast water. 1: Seasonal factors affecting the transport of harmful diatoms and dinoflagellates. Marine Ecology Progress Series 176: 243–251. doi:10.3354/meps176243

Zweng, M.M, Reagan, J.R., Antonov, J.I., Locarnini, R.A., Mishonov, A.V., Boyer, T.P., Garcia, H.E., Baranova, O.K., Johnson, D.R., Seidov, D., and Biddle, M.M. 2013. World Ocean Atlas 2013, Volume 2: Salinity. Edited by S. Levitus and A. Mishonov. NOAA Atlas NESDIS 74: 39 p.

TABLES

Table 1. The D-2 ballast water discharge regulation from the International Convention for the Control and Management of Ships' Ballast Water and Sediments (IMO 2004; Table 1 in Casas-Monroy et al. 2014).

Category	Size Range	Discharge Standard
Phytoplankton	≥ 10 µm - < 50 µm	< 10 cells/mL
Zooplankton	≥ 50 µm	< 10 organisms/m^3
Microbes	*Vibrio cholera*	< 1 CFU per 100 mL or < 1 CFU per 1 g (wet weight) zooplankton samples
	Escherichia coli	< 250 CFU per 100 mL
	Intestinal Enterococci	< 100 CFU per 100 mL

Table 2. Scenarios of management strategies examined in this study.

Management Scenario	Definition
No-Management	The control scenario where neither exchange nor treatment occurred.
Exchange-Only	BWE occurred on all ship-trips. It was assumed that the total concentration of organisms and the proportion that were harmful or non-indigenous did not change pre- versus post-exchange, and that BWE was 100% efficient at purging source port organisms. Therefore, only species belonging to mid-ocean communities were modelled following BWE.
Treatment-Only (100%)	The no-management scenario was modelled with the application of the IMO D-2 standard on 1) 100%, and 2) 50% of voyages. For the second scenario, it was assumed that untreated ballast water was discharged on half of the ship-trips.
Treatment-Only (50%)	
Exchange Plus Treatment (100%)	The exchange-only scenario was modelled with the application of the IMO D-2 standard on 1) 100%, and 2) 50% of trips. For the second scenario, it was assumed that ballast water was managed using only BWE on half of the voyages.
Exchange Plus Treatment (50%)	

Table 3.Shipping pathways examined in this study. See Figure 1 for a map of the geographic boundaries of the Canadian regions.

Shipping Pathway	Definition
Pacific International	Ships destined for ports in British Columbia from foreign source ports.
Atlantic International	Ships destined for ports in Atlantic Canada from foreign source ports. The Atlantic region included the marine ports in the St. Lawrence River and Estuary downstream of (excluding) Québec City, and the four Atlantic provinces, excepting Labrador.
Great Lakes-St. Lawrence River (GLSLR) International	Ships destined for freshwater Canadian ports in the Great Lakes or the St. Lawrence River from foreign source ports. The GLSLR region included Canadian ports upstream of and including Québec City. Transits between American and Canadian ports within this region were not included.
Arctic International	Ships destined to ports in the Canadian Arctic from either foreign source ports or other Canadian regions (domestic transits). The Arctic region included areas delineated by PAME’s 2013 Arctic LME map, including Labrador (PAME 2013).
Arctic Domestic	
All Shipping Pathways	All of the above shipping pathways of interest are combined.

Table 4. Sample size of voyages (n = 2978) and ports for each salinity combination within a given shipping pathway. The sample size of source and recipient ports are the first and second numbers in parentheses, respectively.

Recipient Port Salinity	Source Port Salinity	All Shipping Pathways	Pacific International	Atlantic International	GLSLR International	Arctic International	Arctic Domestic
Fresh	Fresh	93 (28; 13)	9 (6; 2)	17 (7; 2)	67 (15; 9)	0	0
	Brackish	84 (31; 14)	17 (6; 2)	9 (4; 2)	58 (21; 10)	0	0
	Marine	330 (147; 18)	136 (55; 3)	7 (4; 2)	187 (88; 13)	0	0
Brackish	Fresh	21 (6; 4)	0	21 (6; 4)	0	0	0
	Brackish	33 (6; 4)	1 (1; 1)	32 (5; 3)	0	0	0
	Marine	54 (36; 4)	26 (20; 1)	28 (16; 3)	0	0	0
Marine	Fresh	455 (66; 40)	105 (22; 7)	329 (39; 28)	0	4 (4; 3)	17 (1; 2)
	Brackish	638 (62; 28)	153 (17; 7)	478 (41; 18)	0	7 (4; 3)	0
	Marine	1270 (354; 45)	940 (199; 11)	307 (140; 27)	0	19 (13; 5)	4 (2; 2)

Table 5. Number of empirical species abundance distributions for each shipping pathway.

Shipping Pathway	**Number of Species Abundance Distributions**	
	Non-Indigenous Zooplankton	Harmful Phytoplankton
Pacific International	50	45
Atlantic International	39	44
GLSLR International	19	16
Arctic International	31	44
Arctic Domestic	8	44

Table 6. Model sensitivity to changes in input parameters. The response variable is the percentage change in expected number of species establishing per decade (SpPD) among all shipping pathways. To determine the sensitivity of the model, a 25% shift was applied to the ship-trip volume, mean sample density μ, and mean non-indigenous/harmful β parameters, and source ports were randomly assigned to recipient ports within each pathway. Furthermore, the per-propagule probability of establishment (α) and Allee effect (c) were set to 0.05 and 2, respectively, for all species.

Taxonomic Group	**Management Scenario**	**Increase transit frequency (25%)**	**Decrease transit frequency (25%)**	**Randomize source and recipient ports in each pathway**	**Increase mean sample density μ (25%)**	**Decrease mean sample density μ (25%)**	**Increase mean non-indigenous/ harmful β (25%)**	**Decrease mean non-indigenous/ harmful β (25%)**	**$\alpha = 0.05$ (all species)**	**Allee effect ($c = 2$)**
Non-Indigenous Zooplankton	No-Management	3.75%	-8.14%	-2.45%	3.10%	-6.89%	-5.92%	7.17%	2675.10%	103.29%
	Exchange-Only	5.95%	-7.70%	-3.14%	3.09%	-7.33%	-5.12%	5.67%	2665.31%	104.61%
	Treatment-Only (50%)	7.89%	-8.53%	-2.86%	5.44%	-9.18%	-3.21%	9.00%	2782.70%	111.34%
	Exchange Plus Treatment (50%)	8.26%	-6.39%	-1.17%	7.21%	-4.45%	-4.04%	9.61%	2787.93%	111.78%
	Treatment-Only (100%)	9.17%	-25.38%	-10.70%	-3.06%	-3.98%	-9.17%	2.45%	1229.36%	51.38%
	Exchange Plus Treatment (100%)	-3.66%	-25.91%	-15.55%	-13.11%	-11.59%	-16.46%	-4.57%	1057.62%	39.94%
Harmful Phytoplankton	No-Management	3.63%	-8.12%	-5.85%	4.25%	-5.85%	-4.92%	1.78%	2613.72%	92.12%
	Exchange-Only	2.55%	-8.75%	-7.53%	3.28%	-6.68%	-4.80%	0.91%	2595.44%	91.92%
	Treatment-Only (50%)	3.26%	-10.14%	-6.74%	4.04%	-8.58%	-6.10%	1.28%	2691.28%	94.96%
	Exchange Plus Treatment (50%)	3.81%	-8.88%	-6.62%	3.17%	-5.29%	-2.61%	1.27%	2698.45%	95.70%
	Treatment-Only (100%)	4.59%	-20.04%	-8.77%	-8.56%	-5.64%	-9.19%	-3.76%	1801.67%	16.49%
	Exchange Plus Treatment (100%)	9.26%	-18.11%	-5.89%	-6.11%	-5.05%	-7.37%	-0.63%	1914.32%	24.00%

FIGURES

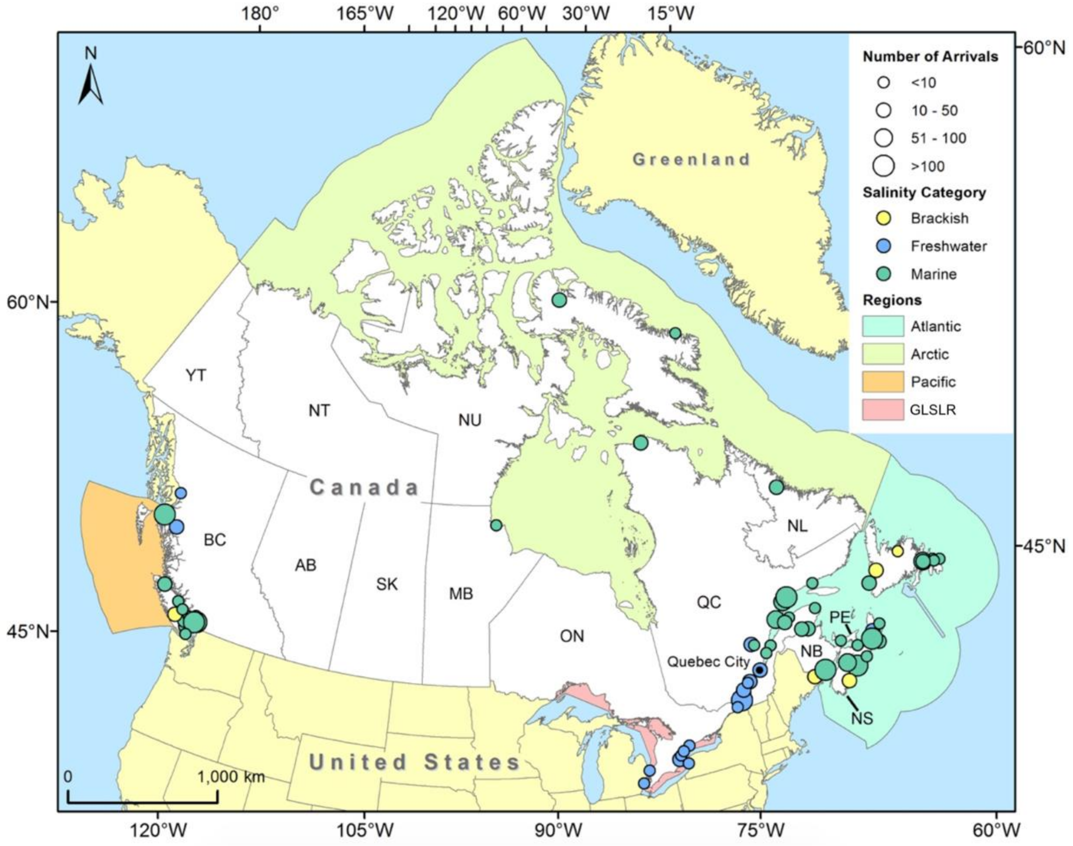


Figure 1. The Canadian geographic regions with the shipping ports examined in this study. The four Canadian regions of interest are the Pacific, Atlantic, Great Lakes-St. Lawrence River (GLSLR), and Arctic. The destination ports (n = 72) included in this study are displayed by the markers where their color and size represent their salinity category and number of ship-trip arrivals, respectively.

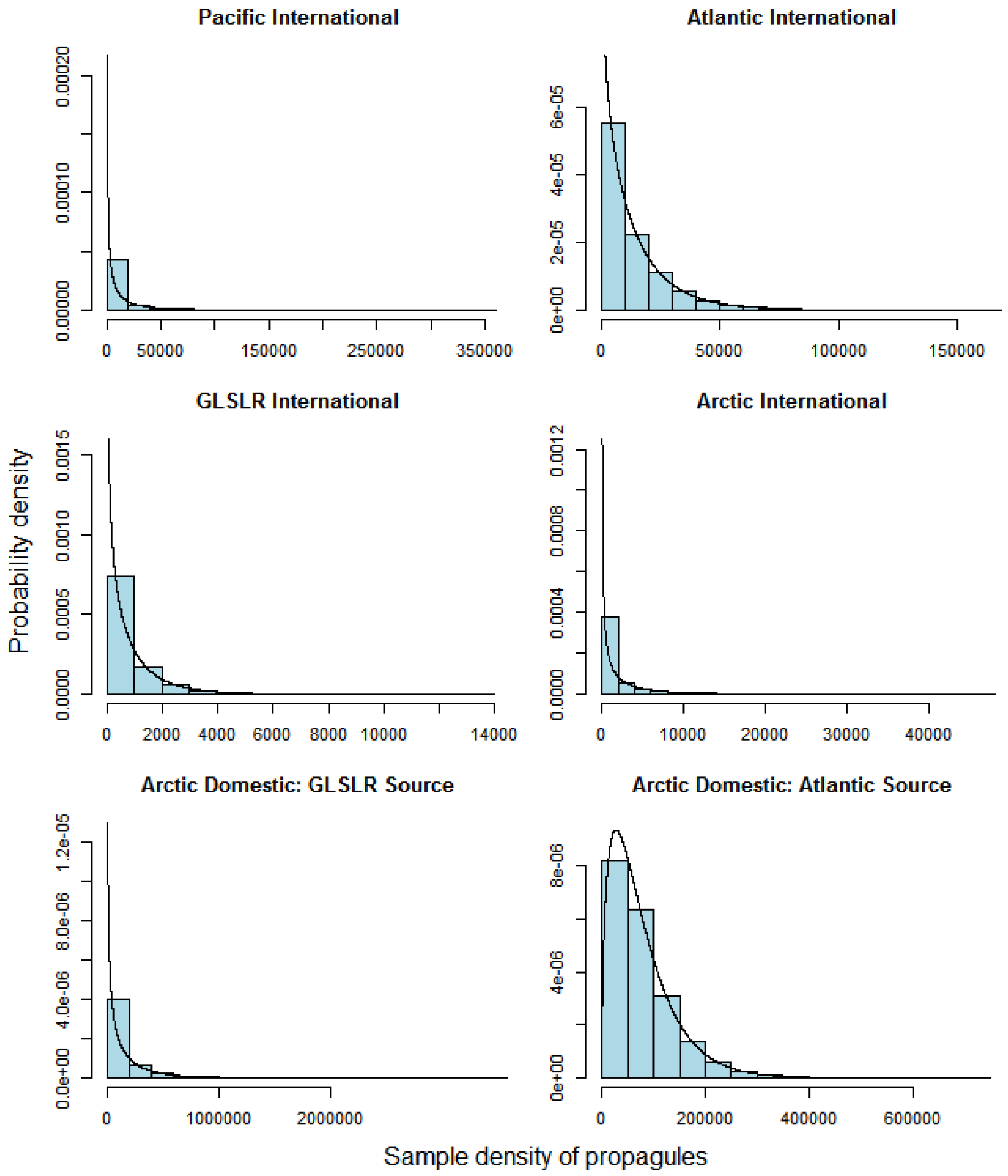


Figure 2. Probability distribution describing the zooplankton sample density among ship-trips within each shipping pathway. The Arctic Domestic pathway used zooplankton data from ships arriving to the Arctic from Atlantic Canada (bottom right panel), and from internal Great Lakes-St. Lawrence River (GLSLR) transits (bottom left panel). The black lines represent the probability density function.

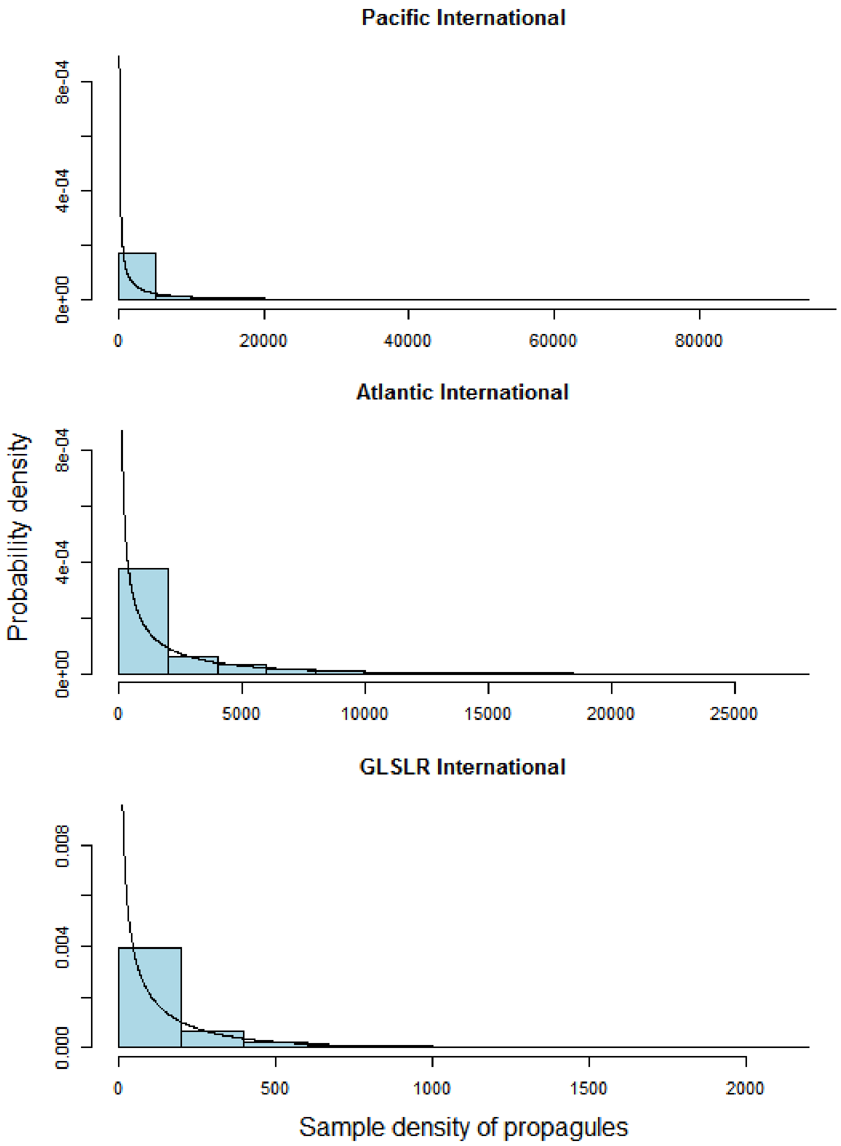


Figure 3. Probability distribution describing the phytoplankton sample density among ship-trips within each shipping pathway. For Arctic International and Arctic Domestic, the phytoplankton sample density distribution was assumed to be equivalent to that of Atlantic International. The black lines represent the probability density function.

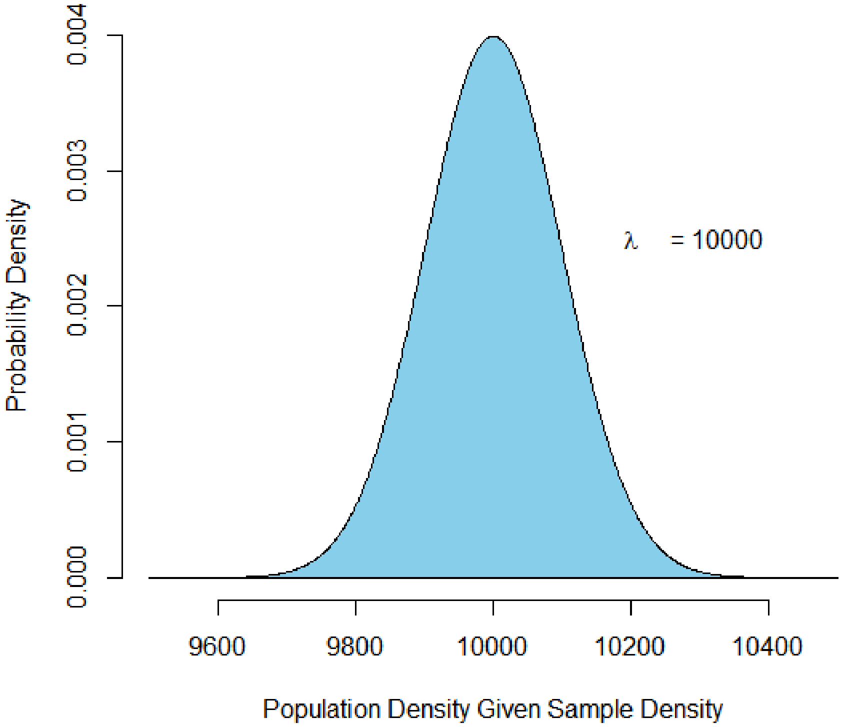


Figure 4. Probability distribution describing the population density of organisms in a single ship (units of organism density/m^3*), given that a sample density of λ=10000 has been obtained. Results shown are from the Pacific International pathway for zooplankton (D.A.R. Drake, Fisheries and Oceans Canada, unpublished data).*

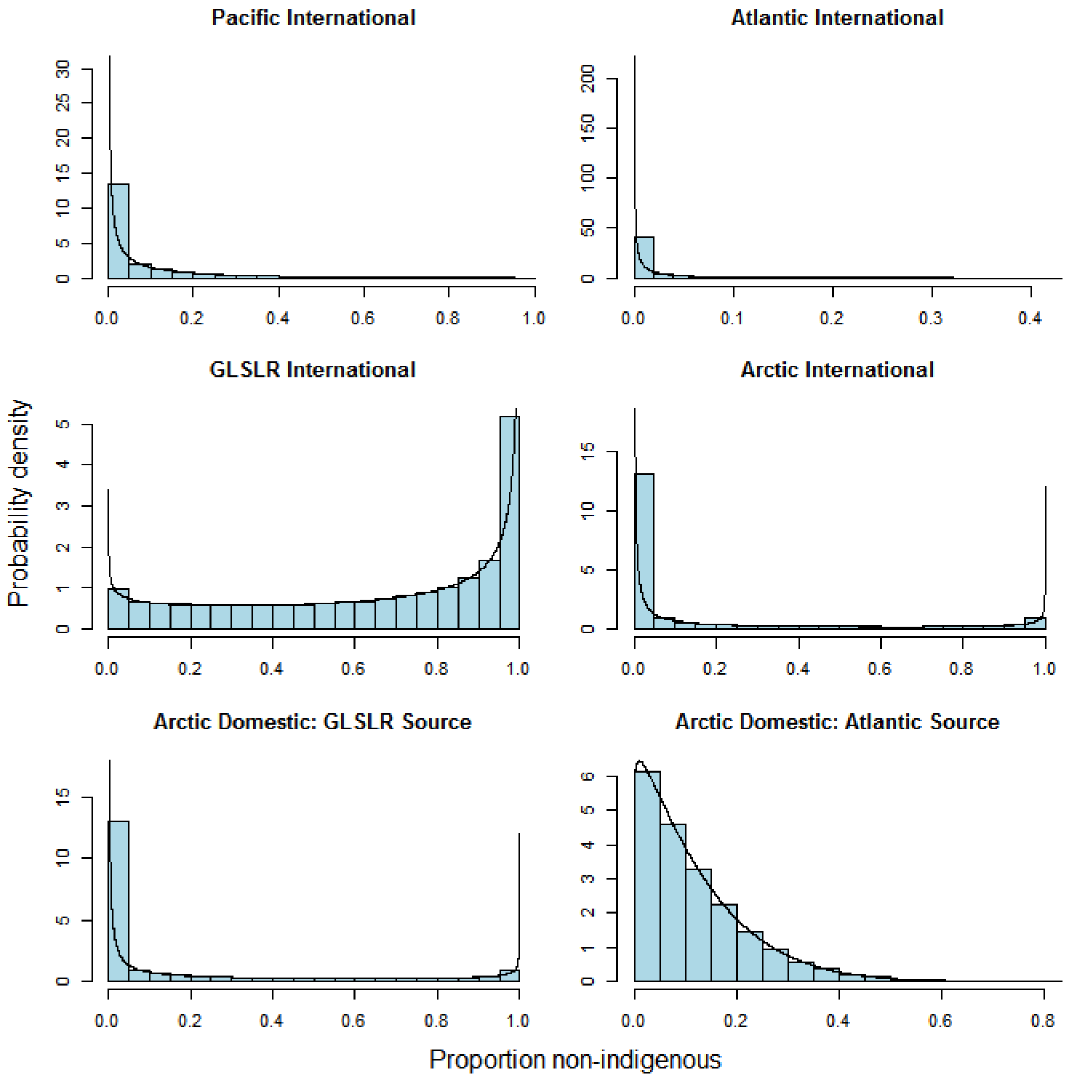


Figure 5. Probability distribution describing the proportion of non-indigenous zooplankton out of the total zooplankton population among ship-trips within each shipping pathway. The Arctic Domestic pathway used zooplankton data from ships arriving to the Arctic from Atlantic Canada (bottom right panel), and from internal Great Lakes-St. Lawrence River (GLSLR) transits (bottom left panel). The black lines represent the probability density function.

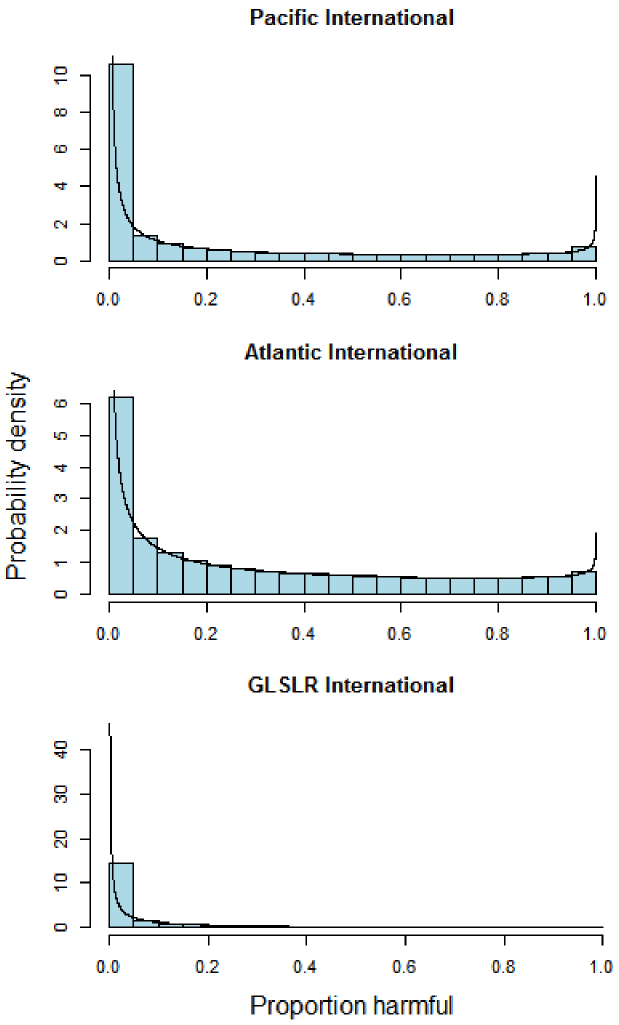


Figure 6. Probability distribution describing the proportion of harmful phytoplankton out of the total phytoplankton population among ship-trips within each shipping pathway. For Arctic International and Arctic Domestic, the probability distribution for the proportion of harmful phytoplankton was assumed to be equivalent to that of Atlantic International. The black lines represent the probability density function.

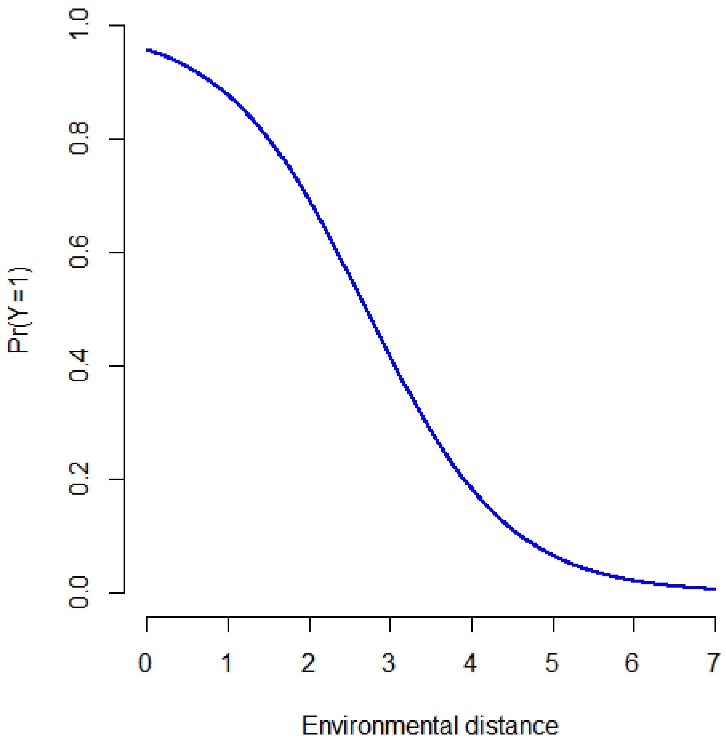


Figure 7. The environmental distance-survival curve. Pr(Y = 1) represents the probability of survival in the recipient environment given the environmental distance which represents the degree of similarity in the temperature and salinity between source and recipient environments. Following Bradie et al. (2015), environmental distance was calculated as Euclidean distance between four environmental variables (minimum, maximum, and mean temperatures, and salinity). The survival curve was fit using a binomial generalized-linear model with data for 603 established aquatic species to determine the relationship between environmental distance and survival.

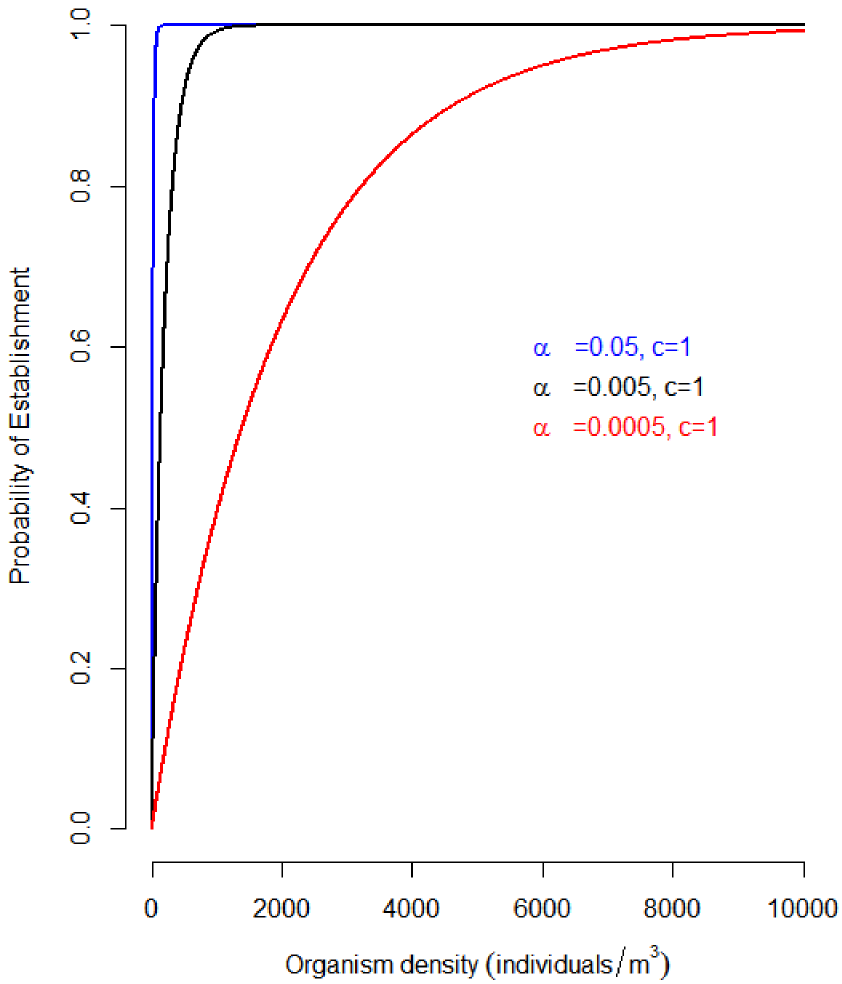


Figure 8. Examples of the probability of establishment based on the per-capita probability of establishment (α), initial population size (N), and Allee effect (c). The probability of establishment was determined using the equation from the National Research Council (2011, Box 4-1).

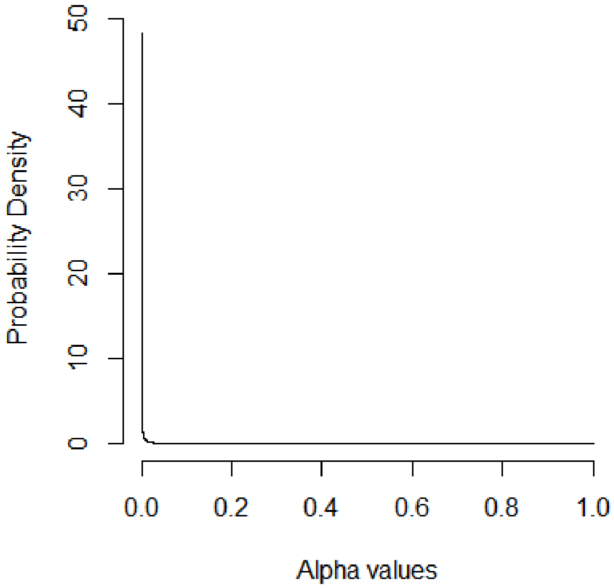


Figure 9. The probability distribution describing the per-capita probability of establishment (α) across multiple species in a ballast tank. This distribution was identical across all trips and pathways.

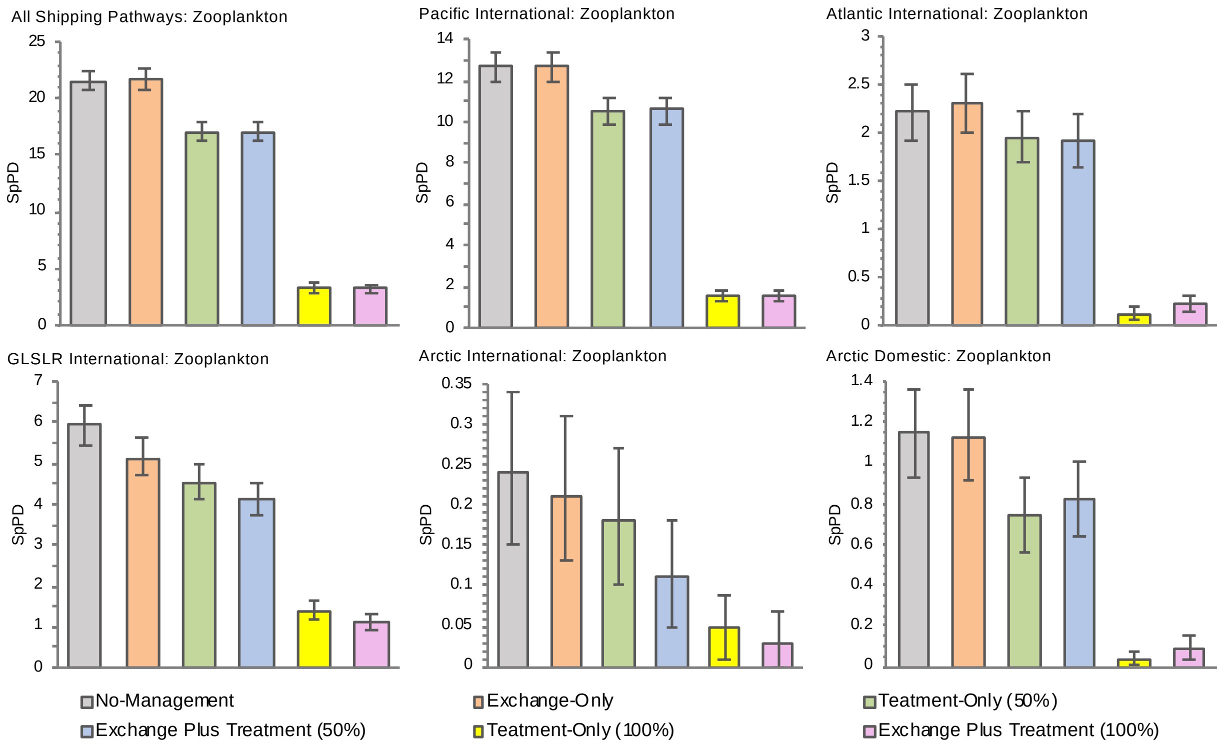


Figure 10. Expected number of species establishments per decade (SpPD) for non-indigenous zooplankton under the different treatment scenarios. Each panel represents a different shipping pathway. The error bars represent the bootstrapped 95% confidence intervals of the mean of SpPD across 1000 years.

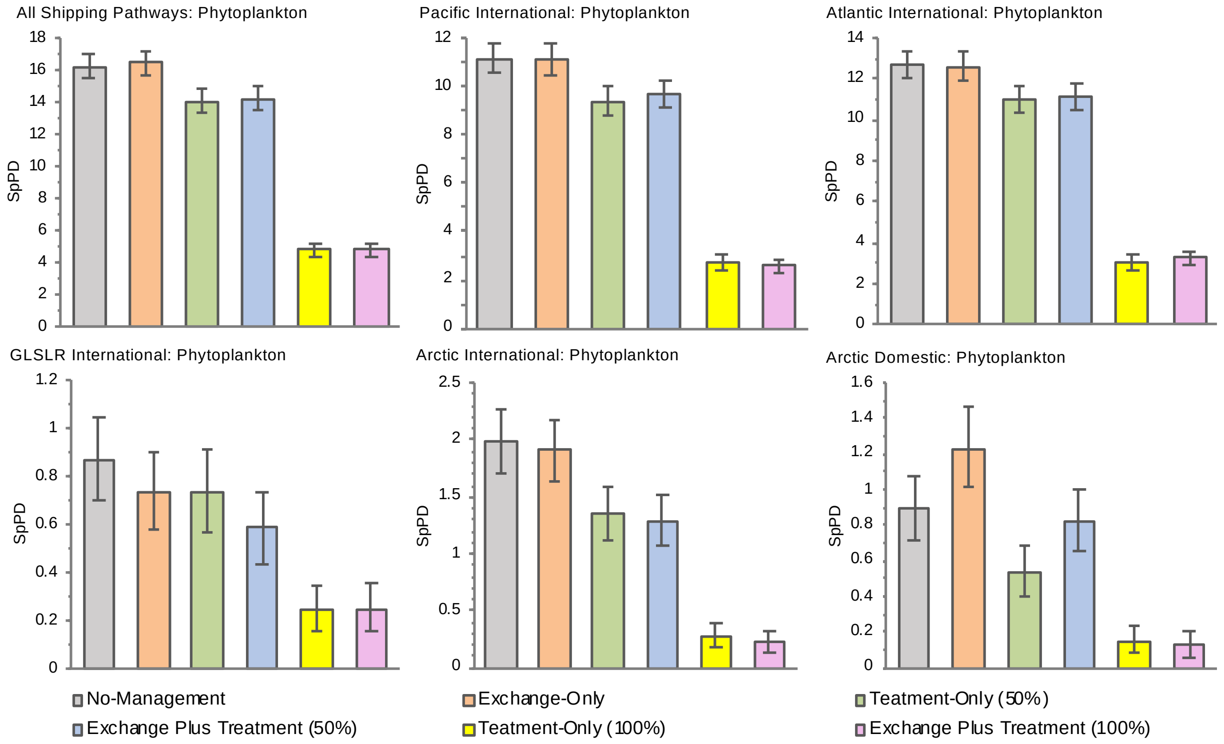


Figure 11. Expected number of species establishments per decade (SpPD) for harmful phytoplankton under different treatment scenarios. Each panel represents a different shipping pathway. The error bars represent the bootstrapped 95% confidence intervals of the mean of SpPD across 1000 years.

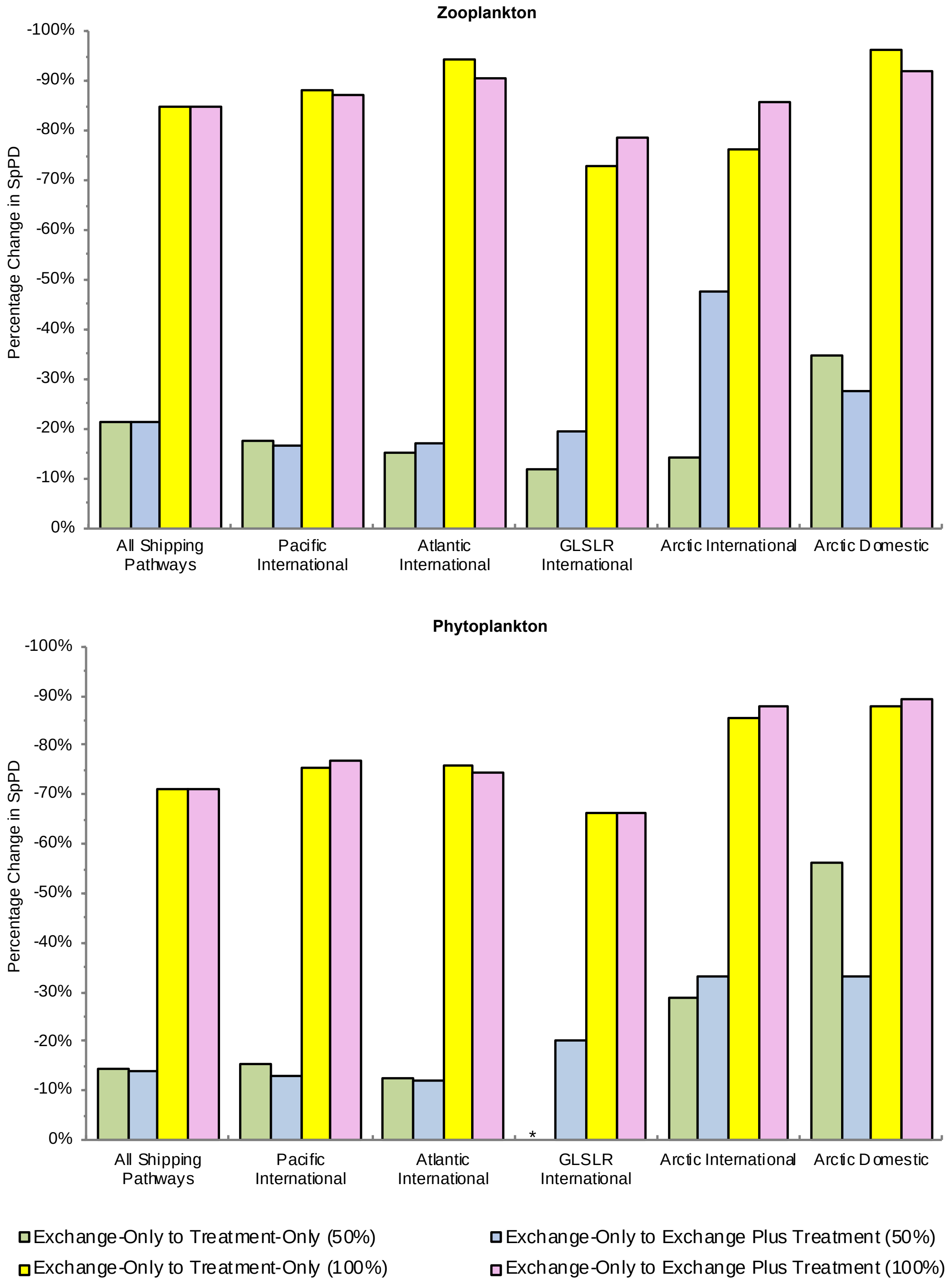


*Figure 12. Percentage change in the expected number of species establishments per decade (SpPD) for non-indigenous zooplankton and harmful phytoplankton for each management scenario compared to that of exchange-only. The management scenarios where there was no change in SpPD from exchange-only is denoted by *.*

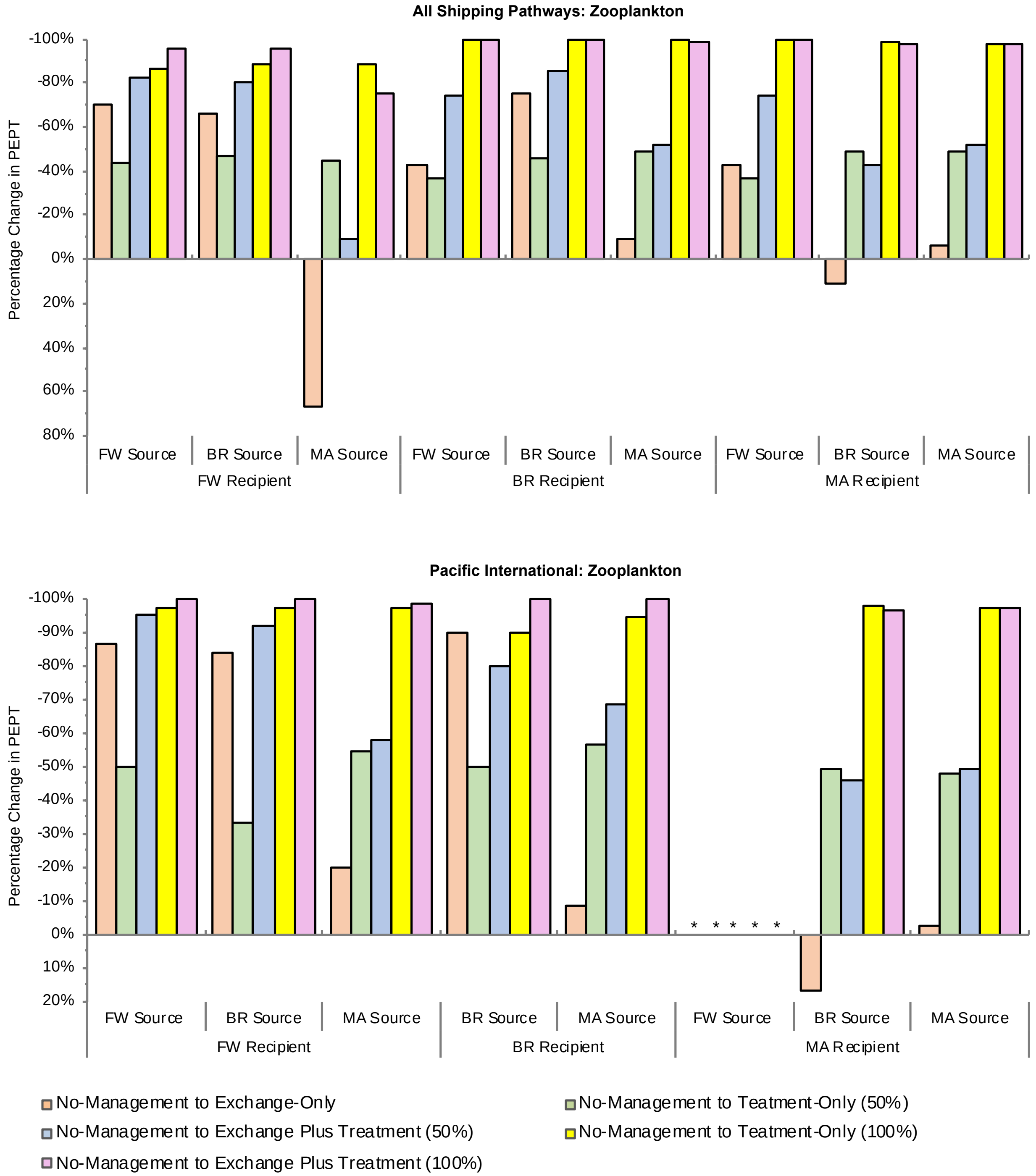


*Figure 13. Percentage change in PEPT (expected probability that at least one species establishes per trip) for non-indigenous zooplankton, for each management scenario, compared to that of the no-management scenario, for the source and recipient port salinity combinations. The salinity categories of fresh, brackish, and marine water are each denoted by FW, BR, and MA, and * denotes when there is no change in PEPT from no-management. Separate panels show different shipping pathways.*

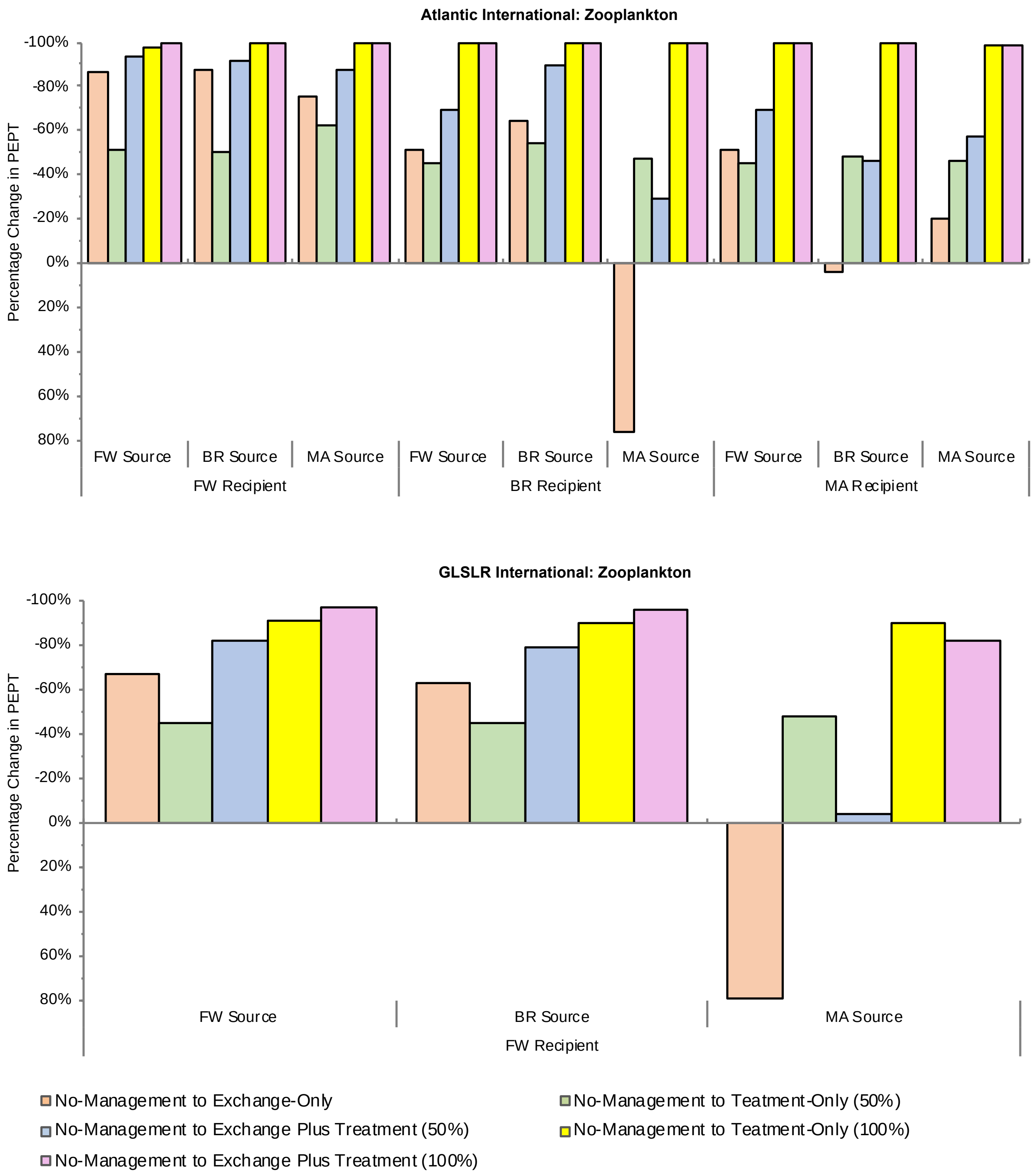


Figure 13. Continued.

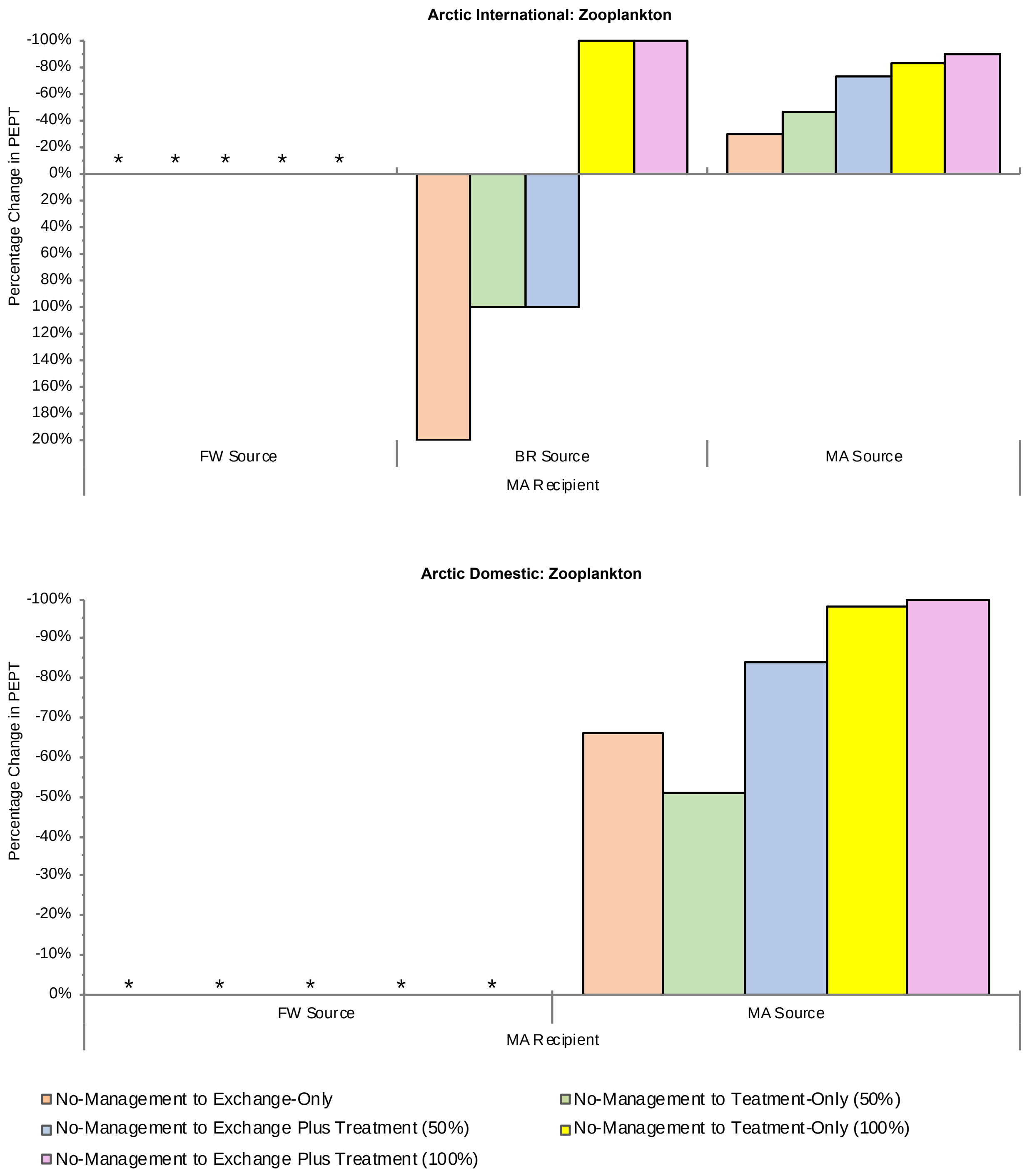


Figure 13. Continued.

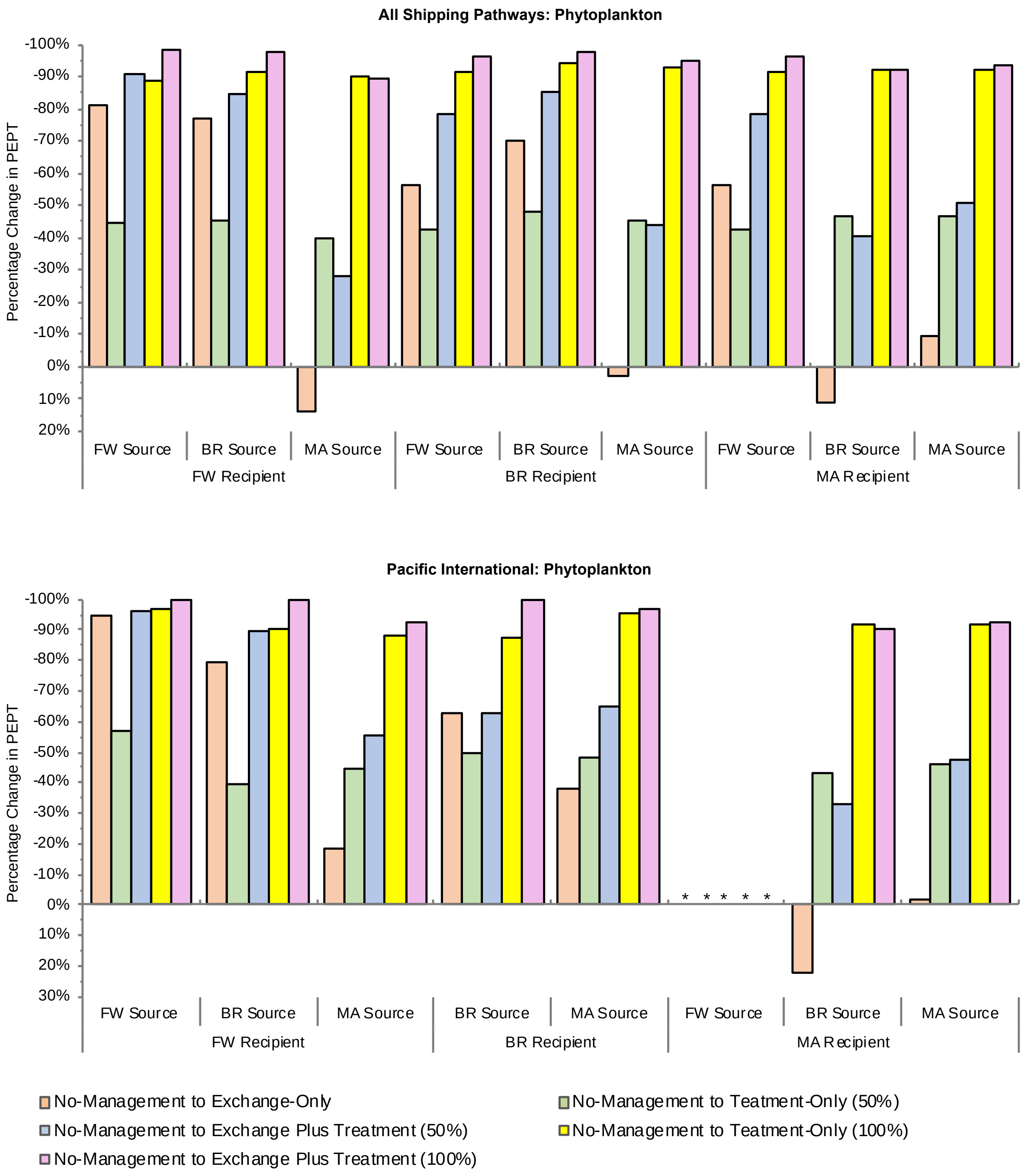


*Figure 14. Percentage change in expected probability that at least one species establishes per trip (PEPT) for harmful phytoplankton, for each management scenario, compared to that of no-management, for the source and recipient port salinity combinations. The salinity categories of fresh, brackish, and marine water are each denoted by FW, BR, and MA, and * denotes when there is no change in PEPT from no-management. Separate panels show different shipping pathways.*

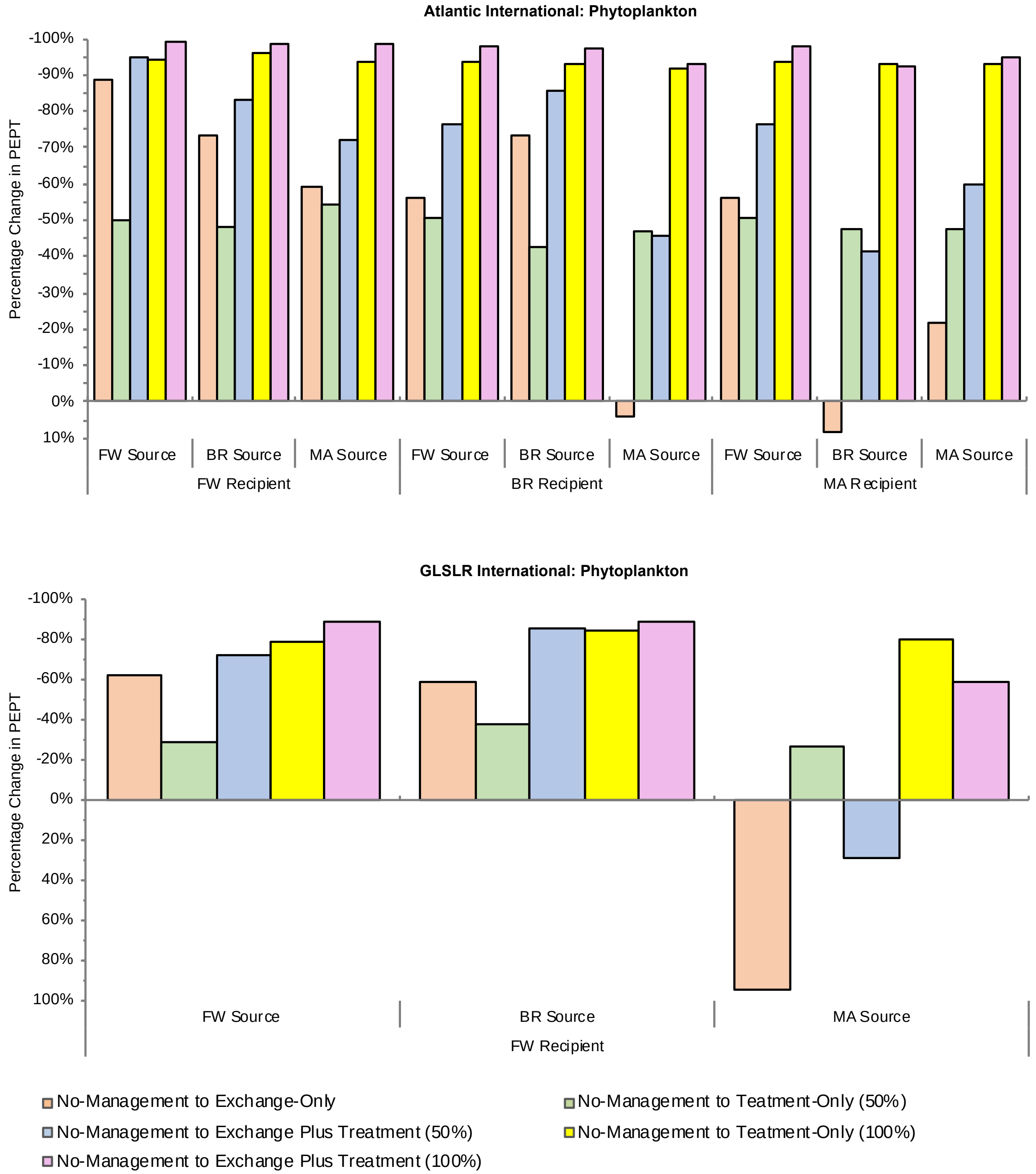


Figure 14. Continued.

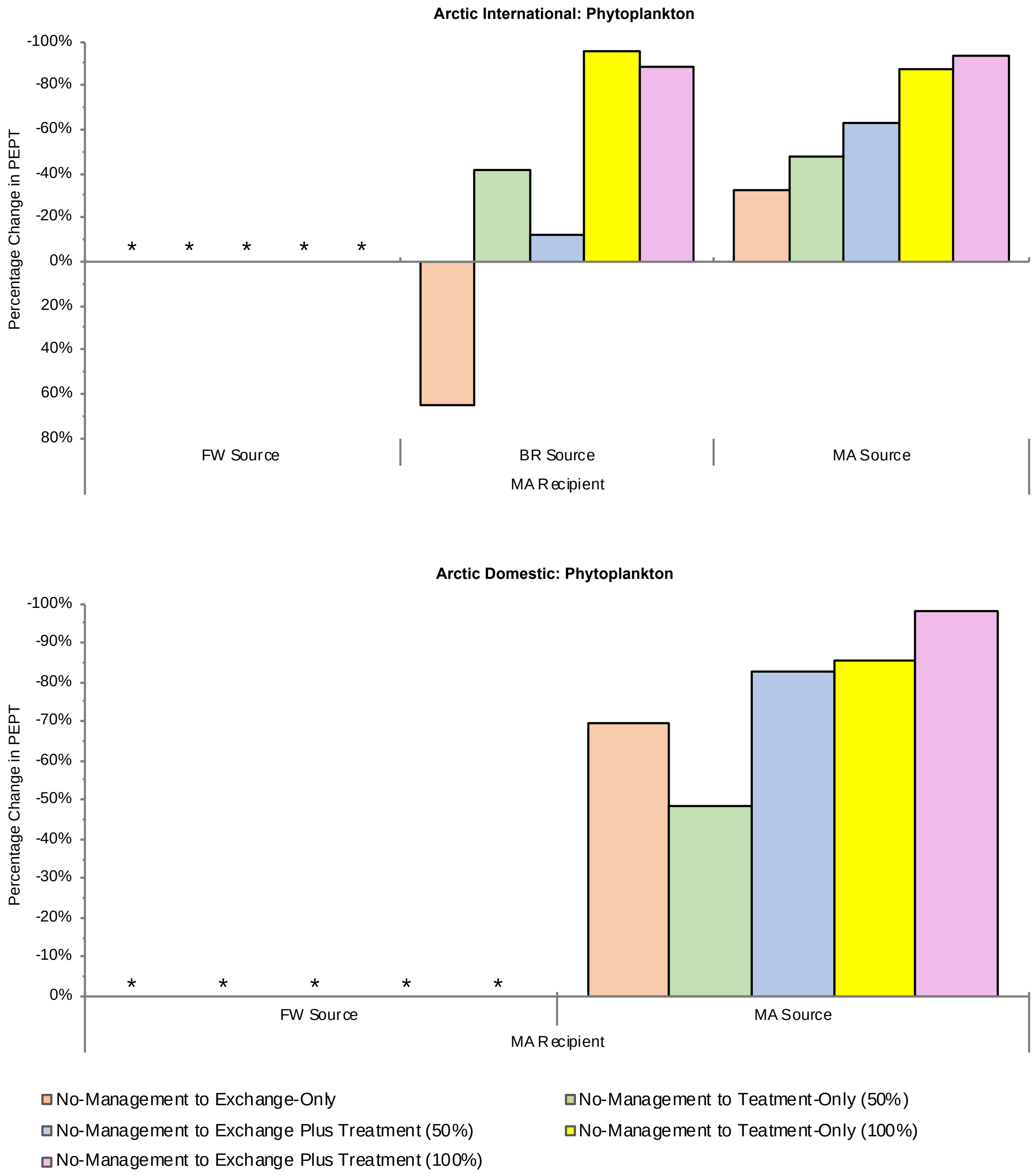
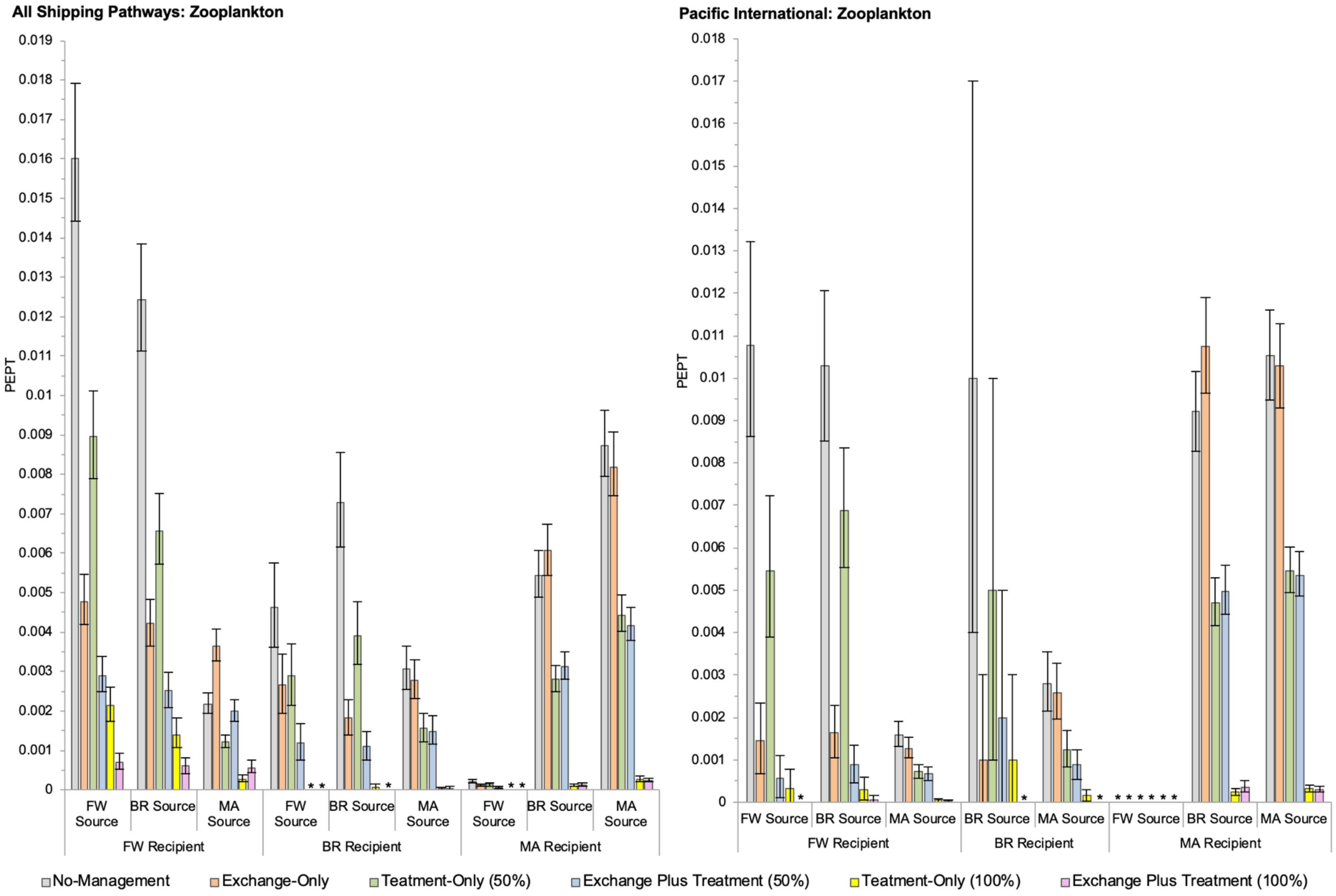


Figure 14. Continued.



All Shipping Pathways: Zooplankton

Pacific International: Zooplankton

*Figure 15. Expected probability that at least one species establishes per trip (PEPT) for non-indigenous zooplankton, for the source and recipient port salinity pairs within each shipping pathway. The salinity categories of fresh, brackish, and marine water are each denoted by FW, BR, and MA. The error bars represent the 95% confidence intervals of the mean of PEPT across 1000 years, and * denotes the scenarios that have zero PEPT. Separate panels show different shipping pathways.*

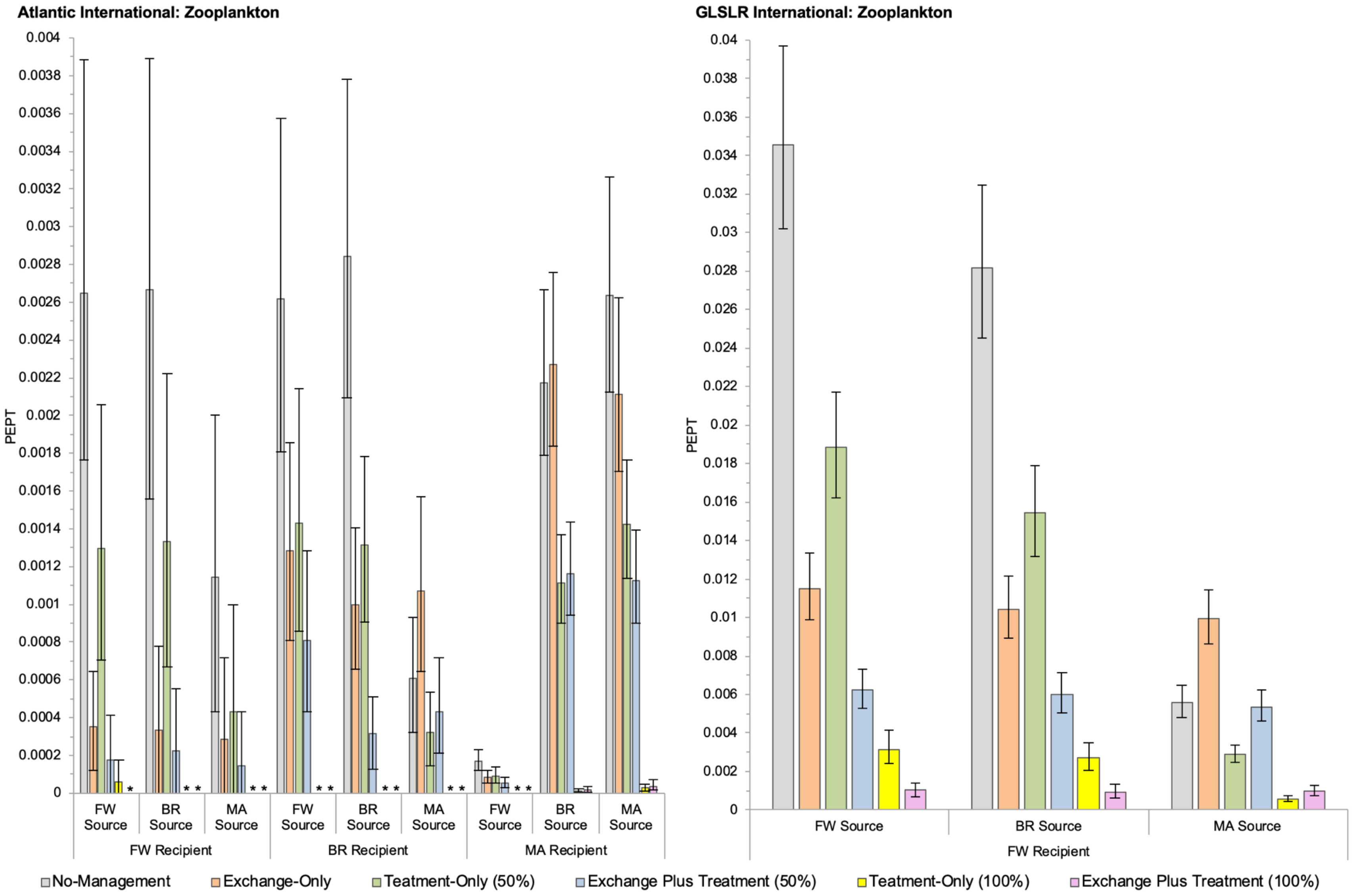


Figure 15. Continued.

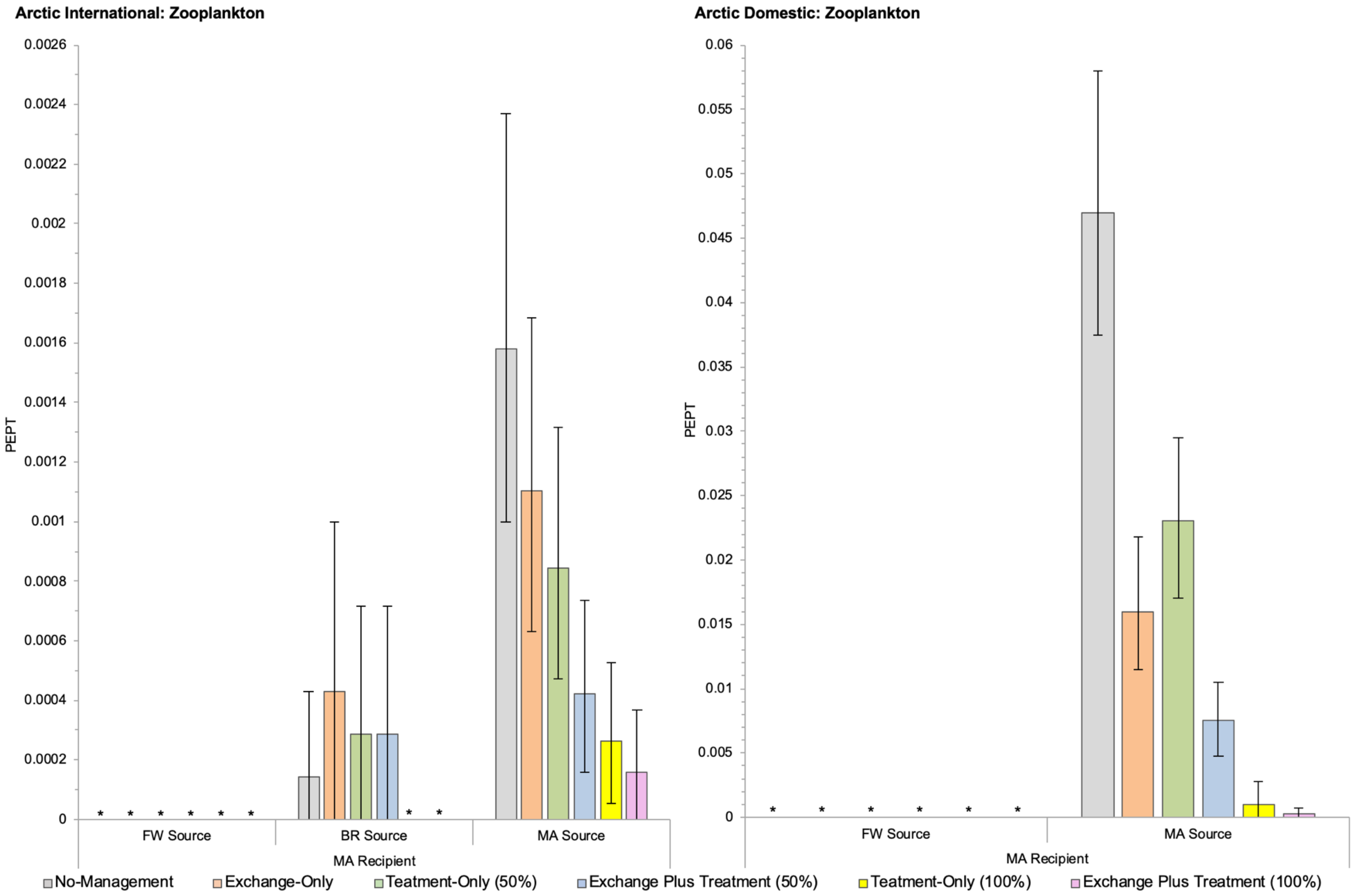


Figure 15. Continued.

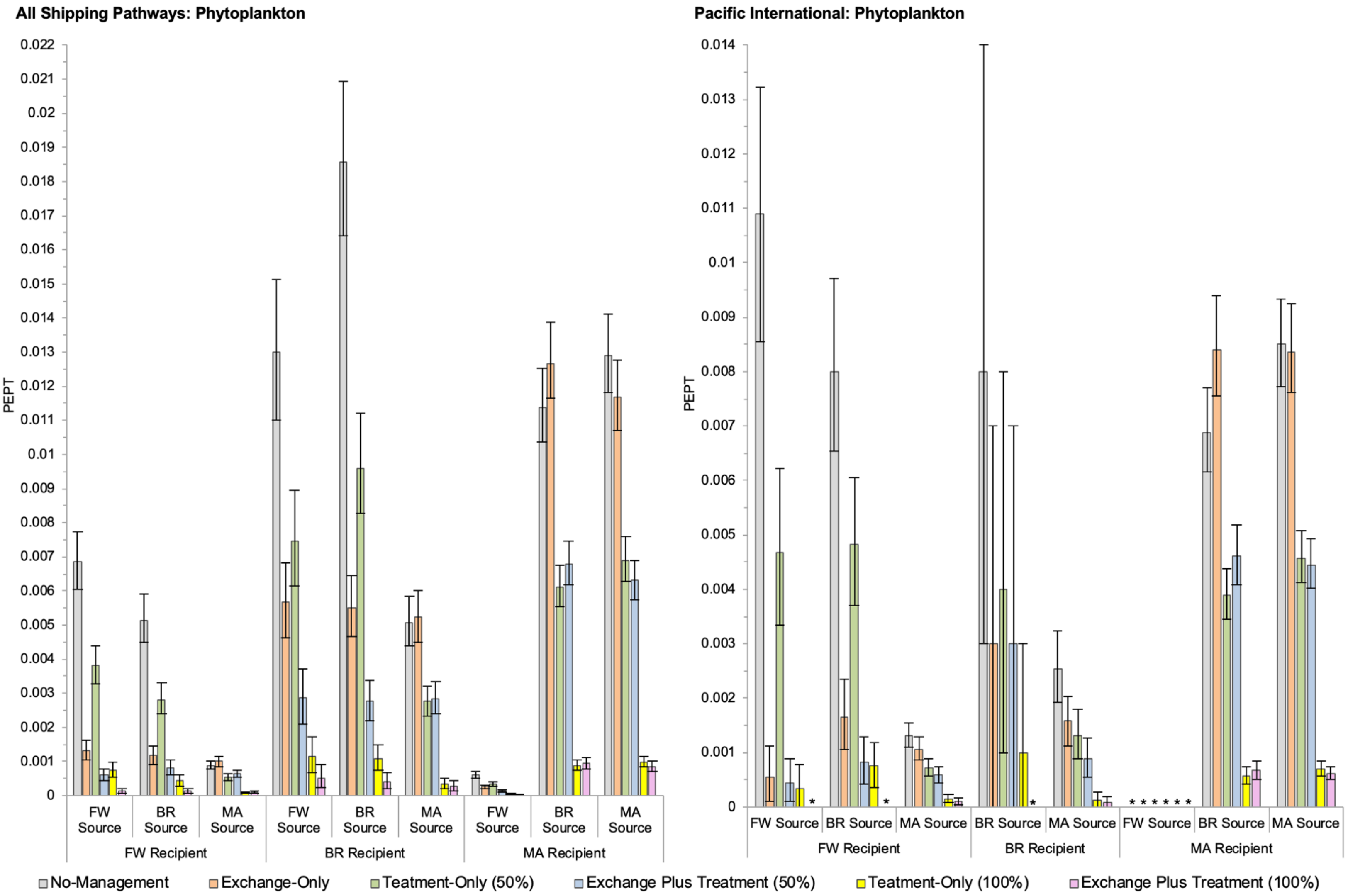


*Figure 16. Expected probability that at least one species establishes per trip (PEPT) for harmful phytoplankton, for the source and recipient port salinity pairs within each shipping pathway. The salinity categories of fresh, brackish, and marine water are each denoted by FW, BR, and MA. The error bars represent the bootstrapped 95% confidence intervals of the mean of PEPT across 1000 years, and * denotes the scenarios that have zero PEPT. Separate panels show different shipping pathways.*

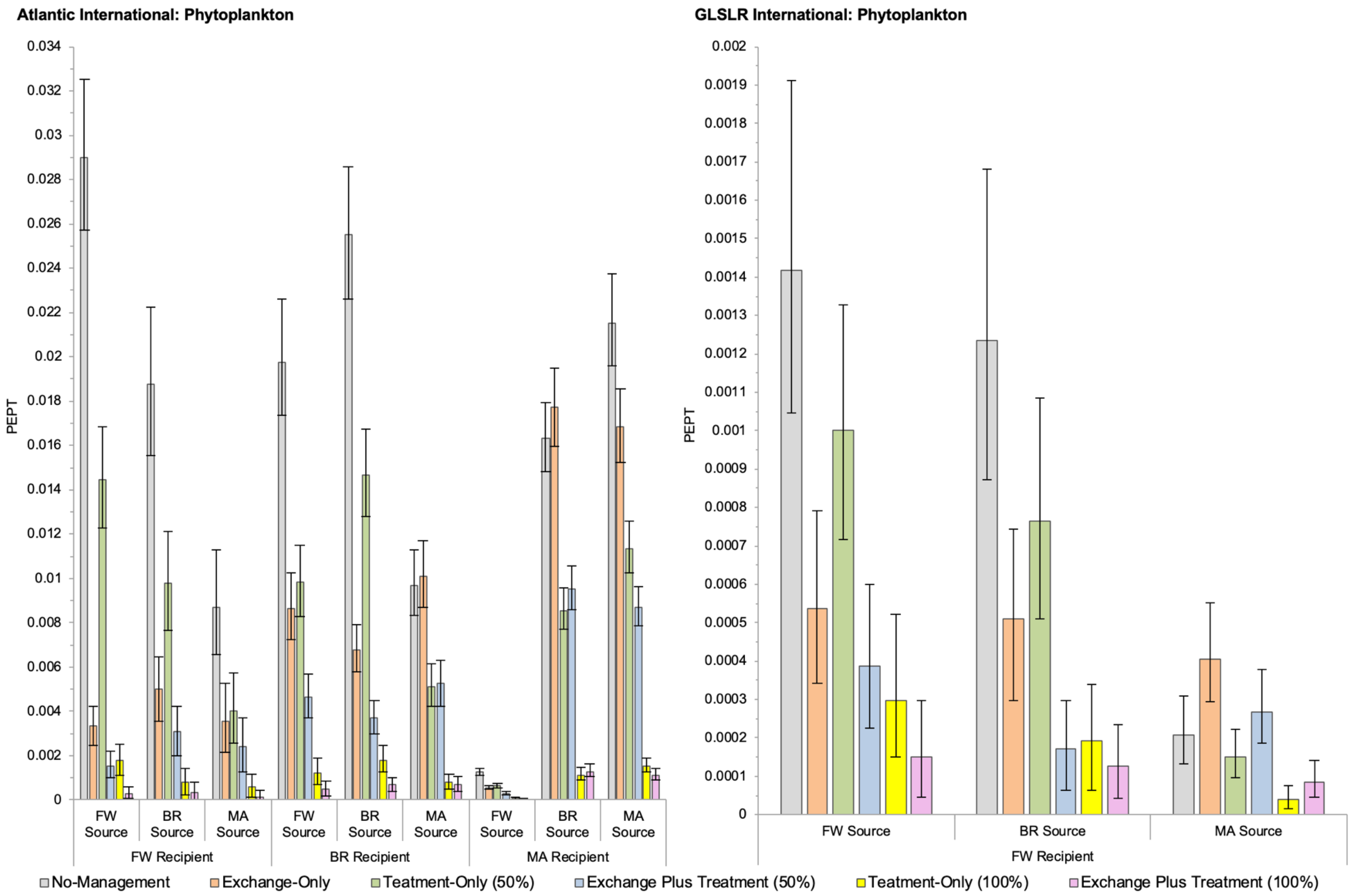


Figure 16. Continued.

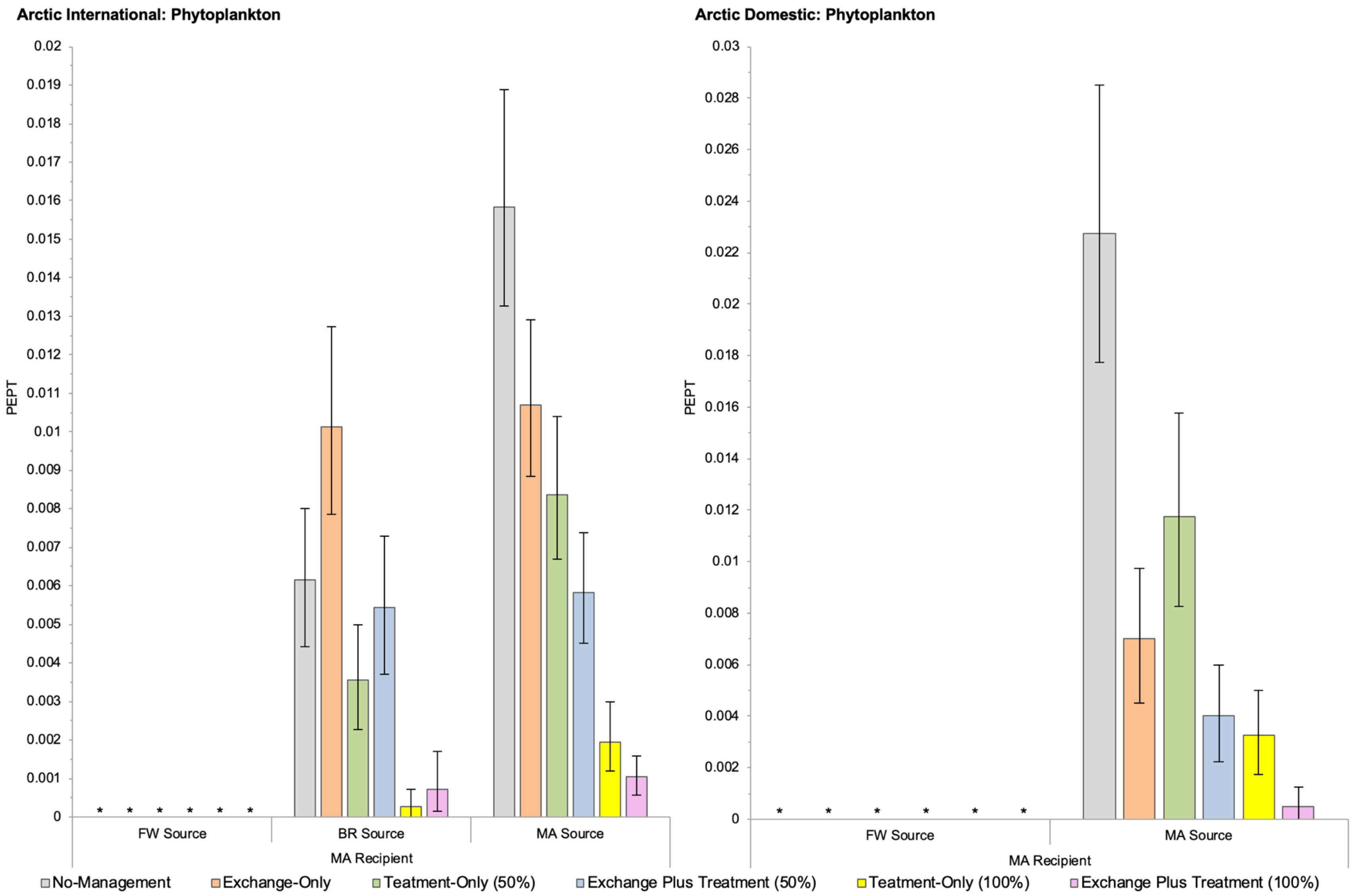


Figure 16. Continued.

APPENDICES

APPENDIX 1

List of Canadian ports and their associated salinity and region. This list is comprised of the Canadian ports (n = 74) used in this study, displaying each of their salinity level, salinity category (fresh, brackish, or marine water), and geographical region (Pacific, Atlantic, Great Lakes-St. Lawrence River (GLSLR), or Arctic). The salinity thresholds used to distinguish fresh, brackish, and marine water were ≤5‰, 5.1-18‰, and ≥18‰, respectively.

Port Name	Region	Salinity (‰)	Salinity Category
Churchill	Arctic	26.33	Marine
Deception Bay	Arctic	31.97	Marine
Edward's Cove	Arctic	32.15	Marine
Milne Inlet	Arctic	29.46	Marine
Qikiqtarjuaq	Arctic	31.11	Marine
Auld's Cove	Atlantic	30.35	Marine
Baie-Comeau	Atlantic	28.1	Marine
Bay Bulls	Atlantic	31.66	Marine
Bay Roberts	Atlantic	31.86	Marine
Bayside	Atlantic	10.61	Brackish
Belledune	Atlantic	28.13	Marine
Botwood	Atlantic	16.24	Brackish
Canso	Atlantic	30.4	Marine
Cape Porcupine	Atlantic	30.35	Marine
Chicoutimi	Atlantic	0	Fresh
Come By Chance	Atlantic	31.6	Marine
Conception Bay	Atlantic	31.86	Marine
Corner Brook	Atlantic	13.63	Brackish
Dalhousie	Atlantic	28.13	Marine
Dartmouth	Atlantic	18.38	Marine
Gaspé	Atlantic	29.22	Marine
Gros-Cacouna	Atlantic	25.4	Marine
Halifax	Atlantic	22.85	Marine
Hantsport	Atlantic	23.89	Marine
Havre-Saint-Pierre	Atlantic	30.61	Marine
Les Méchins	Atlantic	29.52	Marine
Little Narrows	Atlantic	2.63	Fresh
Liverpool	Atlantic	16.07	Brackish
Long Harbour	Atlantic	31.64	Marine
Lower Cove	Atlantic	31.46	Marine
Matane	Atlantic	28.55	Marine
Mulgrave	Atlantic	30.35	Marine
Pictou	Atlantic	29.79	Marine

Port Name	Region	Salinity (‰)	Salinity Category
Point Tupper	Atlantic	30.35	Marine
Pointe-au-Pic	Atlantic	22.5	Marine
Port Alfred	Atlantic	26.18	Marine
Port-Cartier	Atlantic	30.08	Marine
Saint John	Atlantic	32.13	Marine
Sept-Îles	Atlantic	30.32	Marine
Sheet Harbour	Atlantic	31.12	Marine
St. John's	Atlantic	31.77	Marine
Summerside	Atlantic	28.65	Marine
Sydney	Atlantic	21.25	Marine
Whiffen Head	Atlantic	31.6	Marine
Bécancour	GLSLR	0	Fresh
Contrecoeur	GLSLR	0	Fresh
Hamilton	GLSLR	0	Fresh
Montréal	GLSLR	0	Fresh
Oakville	GLSLR	0	Fresh
Oshawa	GLSLR	0	Fresh
Port Colborne	GLSLR	0	Fresh
Québec City	GLSLR	0	Fresh
Sarnia	GLSLR	0	Fresh
Sorel	GLSLR	0	Fresh
Toronto	GLSLR	0	Fresh
Trois-Rivieres	GLSLR	0	Fresh
Valleyfield	GLSLR	0	Fresh
Windsor	GLSLR	0	Fresh
Campbell River	Pacific	27.24	Marine
Cowichan Bay	Pacific	26.3	Marine
Crofton	Pacific	26.3	Marine
Fraser Port	Pacific	0	Fresh
Kitimat	Pacific	0.79	Fresh
Nanaimo	Pacific	25.74	Marine
Port Alberni	Pacific	6.09	Brackish
Port McNeill	Pacific	30.94	Marine
Port Mellon	Pacific	24.25	Marine
Prince Rupert	Pacific	29.66	Marine
Sechelt	Pacific	24.8	Marine
Squamish	Pacific	24.25	Marine
Stewart	Pacific	0	Fresh
Texada Island	Pacific	26.47	Marine
Vancouver	Pacific	25.3	Marine
Victoria	Pacific	28.51	Marine

APPENDIX 2

*Model parameters for the agent-based model used to quantify the expected number of non-indigenous zooplankton and harmful phytoplankton species establishing in Canadian ecosystems. The Arctic Domestic pathway used zooplankton data from ships arriving to the Arctic from Atlantic Canada, and from internal Great Lakes-St. Lawrence River (GLSLR) transits. * means the parameter was assumed to be equal to that of the Atlantic International pathway.*

Model Parameter		**Shipping Pathway**					
		GLSLR International	Pacific International	Atlantic International	Arctic International	Arctic Domestic	
						Atlantic	GLSLR
Sample Year		2006	2008	2006	2015	2015	2015
Trips (n; 2978 total)		312	1387	1228	30	4	17
Zooplankton Sample Density (Negative Binomial Distribution)	*size*	0.6297	0.2783	0.8268	0.2894	1.5618	0.4034
	μ	752.00	8861.66	13099.23	1661.77	77349.9	123550.7
Zooplankton Proportion Non-Indigenous (Beta)	α	0.7515	0.2302	0.1842	0.0973	1.0696	0.2411
	β	0.4004	2.9896	14.1509	0.4625	7.9209	1.1468
Phytoplankton Sample Density (Negative Binomial)	*size*	0.3098	0.1299	0.2489	*	*	*
	μ	25.1875	67.8222	228.296	*	*	*
Phytoplankton Proportion Harmful (Beta)	α	0.1934	0.1190	0.2652	*	*	*
	β	1.7654	0.6119	1.4714	*	*	*

All Trips

Parameter		**Value**
Population Density Error		Poisson
Survival Probability (Logistic, Environmental Distance)		Intercept= 3.122, slope = -1.152
Probability Single Propagule Establishes (Beta)	α	0.005
	β	5
Allee Effect	c	1

APPENDIX 3

Supplementary Figure A1

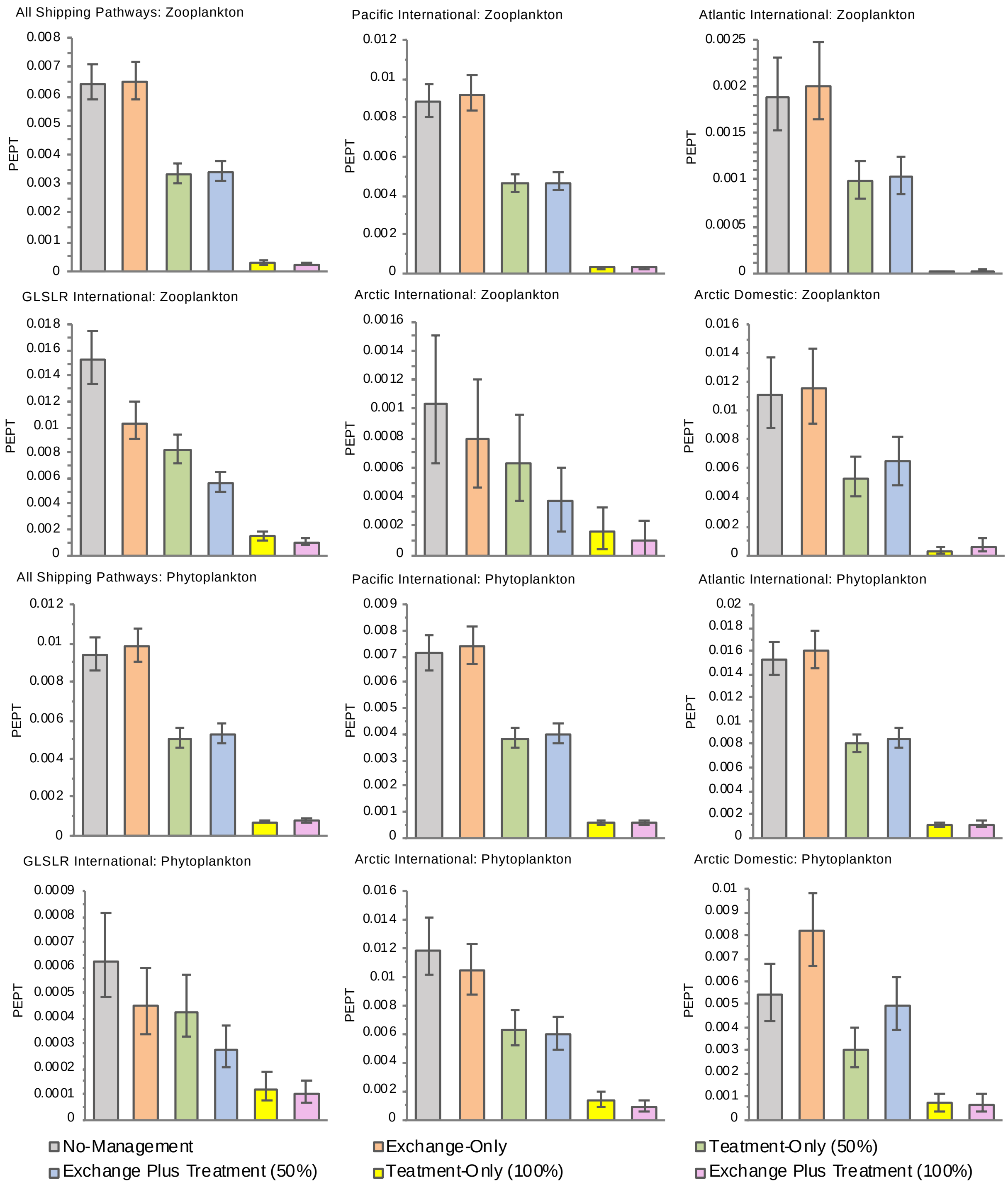


Figure A1. Expected probability that at least one species establishes per trip (PEPT) for non-indigenous zooplankton and harmful phytoplankton within each shipping pathway. The error bars represent the bootstrapped 95% confidence intervals of the mean of PEPT across 1000 years.