INTERNATIONAL MARITIME ORGANIZATION

E

MARINE ENVIRONMENT PROTECTION COMMITTEE
72nd Session
Agenda item 4

MEPC 72/INF.7
8 January 2018
ENGLISH ONLY

HARMFUL AQUATIC ORGANISMS IN BALLAST WATER

An assessment of ballast water treatment to protect Arctic waters

Submitted by FOEI

SUMMARY

Executive summary:	This document offers in the annex a comprehensive analysis of shipboard ballast water management system performance, based on available data
Strategic direction:	2
High-level action:	
Output:	No related provisions
Action to be taken:	Paragraph 2
Related document:	MEPC 69/INF.17

Introduction

1 Friends of the Earth U.S. recently commissioned a report to evaluate shipboard ballast water treatment performance, because comparative data has not been assembled in a single document, thereby hindering informed policy-making and oversight at IMO and other relevant fora. The main finding of the report is that treatment system performance varies a great deal, both across the different treatment types and within the most common treatment types, with some devices outperforming others by many orders of magnitude. For example, with regard to reduction of phytoplankton concentrations, the best system outperformed the worst by over one million times, on average. The report is set out in the annex.

Action requested of the Committee

2 The Committee is invited to note the information provided.

***

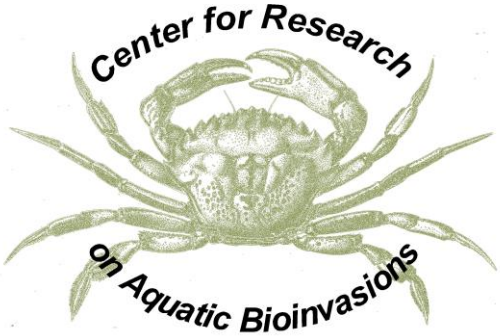
IMO OUR HERITAGE BETTER SHIPPING FOR A BETTER FUTURE 70

ANNEX

An Assessment of Ballast Water Treatment to Protect Arctic Waters

A Report for Friends of the Earth U.S.

Andrew N. Cohen, PhD
Center for Research on Aquatic Bioinvasions
Richmond, California



December 2017

Contents

The Problem 1
- Background on Ballast Water Treatment and Testing
- Boundaries and Temperatures of Arctic Marine Waters
- Source Areas and Invasion Risk
- Treatment Modes

The Analysis 6
- Data Acquisition
- Performance Assessment

Results 9
- Zooplankton
- Phytoplankton
- Bacteria
- Treatment Processes and Low Temperature
- Low Temperature Limits on Treatment System Operation

Comments and Recommendations 31
- Summary of Results and Analysis
- Treatment System Testing; and Access to Performance Data
- Treatment System Performance

Acknowledgments 34

Literature Cited 35

Appendix A. Supporting Analyses 40

Appendix B. Treatment Systems and Sources of Data ..... (*available on request*)

The Problem

The transport of aquatic organisms to waters outside of their native regions has harmed ecosystems, disrupted economic activities, and sickened and sometimes killed people (Carlton 1985; Epstein et al. 1993; McCarthy & Khambaty 1994; Hallegraeff 1998). Compared to warmer regions, few non-native species have become established in the marine waters of the Arctic (Cohen 2016). But warming waters, retreating sea ice, and other climate change-associated effects could quickly change that. Key factors that could dramatically raise the rate of invasion include:

- changing environmental conditions that make Arctic waters less suitable for its current flora and fauna (Fabry 2009; Steinacher *et al.* 2009; Christiansen & Reist 2013), thereby reducing biotic resistance to invasion, even while
- these waters become more suitable for organisms from southern regions, increasing the probability that some will become established (*e.g.* Hegseth & Sundfjord 2008; Renaud *et al.* 2015; Ware *et al*. 2015);[1] meanwhile
- ice retreat opens sea lanes that allow major increases in ship traffic through and into the Arctic, and into new regions of the Arctic (Khon *et al.* 2010; Smith & Stephenson 2013; Miller 2014), inadvertently carrying many more non-native organisms into these waters; in addition
- warming conditions and ice retreat promote an increase in economic activities within the region (including oil and gas exploration and extraction, mining, fishing, and tourism) (Chan *et al.* 2012; Michel 2013; Miller 2014), bringing in more shipping with associated transport of organisms; and in concert with this,
- alteration of nearshore conditions at some sites (construction of harbor facilities, local pollution) creates habitats where invading organisms can more readily gain a toehold (*e.g.* Glasby *et al.* 2007, Kerckhof *et al.* 2011).

Though most of the Earth's bioregions have been subject to anthropogenic biological invasions for centuries or millennia, the pace of aquatic invasions has increased substantially in recent decades (Cohen & Carlton 1998; Ricciardi 2006). Given the increase in invasion risk posed by these combined factors, and the relatively pristine nature of its waters, the future of the Arctic marine region could conceivably include the most rapid invasion by non-native species of any major biogeographic zone in the planet's history.

Aquatic invaders will arrive in the Arctic both as biofouling on vessels, drilling rigs, etc., and in ships' ballast water. Programs to reduce the rate of invasion via the latter pathway rely mainly on shipboard treatment of ballast water. This paper discusses what we know about the ability of shipboard treatment systems to prevent biological invasions into Arctic waters via ballast water, based on an assessment of the available test data for these systems.

[1] Ekman (1953, at p. 184) noted that a common feature "in the high arctic and to a lesser degree in the low-arctic regions is the almost complete lack, or at least the great dearth, of sessile animals within the tidal zone...mainly due to the destructive influence of the drift ice." Thus, with the disappearance of sea ice, the entire intertidal zone could become susceptible to invasions by intertidal organisms from ice-free regions.

Background on Ballast Water Treatment and Testing

Ships load ballast water primarily to reduce buoyancy and increase stability when they are empty or carry little cargo, and primarily discharge ballast water when (or prior to) loading cargo. Ballast water may also be loaded or discharged to adjust trim, to improve maneuverability, to increase propulsion efficiency, or for other navigation-related purposes. Cargo ships carry most of the world's ballast water, though other types of vessels may carry smaller quantities. Typically, even when a cargo ship pumps out its ballast tanks and reports "no ballast on board" ("NOBOB"), some residual or unpumpable ballast, comprising perhaps tens or even hundreds of thousands of gallons, remains in its ballast tanks (Locke *et al.* 1991). Thus most cargo ships carry some ballast water virtually all the time (Carlton *et al.* 1995).

It has been known for over a century that ballast discharges can introduce non-native species into new regions (Ostenfeld 1908). However, this didn't become a general concern until a series of high profile bioinvasions in the 1980s and 1990s, believed to have resulted from ballast discharges, brought the issue to public attention (Nichols *et al.* 1990; Hallegraeff & Bolch 1992; Harbison & Volvik 1994; McCarthy & Khambaty, 1994; Darrigran & Pastorino 1995; Nalepa & Schloesser 2014). In 1991 the UN's International Maritime Organization (IMO) adopted voluntary guidelines to reduce the risk of transporting non-native species in ballast water (IMO 1991), and in 2004 the IMO drafted an international treaty, the Convention for the Control and Management of Ships' Ballast Water and Sediments, which limits the allowable concentrations in ballast water discharges of two organism groups and three indicator bacteria (IMO 2004). The two organism groups are based on size: the larger group comprises organisms whose minimum dimension is greater than or equal to 50 micrometers (μm) and mainly consists of zooplankton, while the smaller group comprises organisms whose minimum dimension is greater than or equal to 10 μm and less than 50 μm and mainly consists of phytoplankton (hereafter referred to as the zooplankton and phytoplankton groups, respectively). Most ships plan to meet these limits through the installation and operation of shipboard ballast water treatment systems.

IMO requires that shipboard treatment systems be tested and certified as capable of meeting these discharge limits, a process known as type approval. The tests are conducted in accordance with IMO guidelines that address some aspects of treatment system testing but not other key elements (SAB 2011; Cohen & Dobbs 2015). The laboratories conducting the tests, which are typically hired and paid by the manufacturers of the treatment systems being tested, are apparently allowed substantial leeway to interpret or modify the specifications that the guidelines do provide. The test results and test reports are then given to the manufacturer, and the manufacturer can use them to apply for type approval. Type approvals are granted by the IMO Member State or Member States to which the manufacturer applies, not by IMO.

For type approval, the IMO guidelines require results from 10 successful trials conducted in a land-based test facility plus 3 successful trials conducted with a treatment system installed on a ship.[2] In each of these trials, the water used to test the treatment system is split into treatment

[2] IMO initially adopted guidelines for type approving ballast water treatment systems, including protocols for conducting the tests needed for type approval, in 2005; adopted amended guidelines in 2008 and 2016; and in 2017 approved a draft "code for approval" that is nearly identical to the 2016 guidelines, and is expected to be adopted in 2018 (IMO 2005, 2008a, 2016, 2017). I'm told that IMO guidelines are considered voluntary and IMO codes are considered mandatory. The 2005 and 2008 guidelines are very similar, but there are some significant differences between these and the 2016 guidelines. The treatment system tests that produced the data analyzed in this paper were all conducted prior to IMO's October 2016 adoption of the 2016 guidelines, and thus were governed by the 2005 or 2008 guidelines. Unless otherwise stated, the descriptions of IMO guidelines in this paper refer to the 2008 guidelines.

and control streams. The control stream is passed into either an actual ballast tank (in shipboard tests) or a large tank intended to simulate a ballast tank (in land-based tests), where it is held for a period of time (variable in shipboard tests; at least 5 days in land-based tests) before being discharged.[3] The procedure for the treatment stream is identical except that the treatment being tested is applied at the appropriate stage or stages, either on intake before entering the tank, while the water is in the tank, or during discharge from the tank. Concentrations of the two organism groups and three indicator bacteria targeted by the regulations are measured at various points, including the points of discharge for both the control and treatment tanks. IMO protocols also require test laboratories to measure the concentration of heterotrophic bacteria.[4]

The guidelines do not require that the test reports or test data be submitted to IMO or released to the public.[5] IMO has invited (but hasn't required) Member States to report to IMO on type approvals granted to treatment systems, including test results (IMO 2008b, 2013). Some form of report has been submitted to IMO for most but not all type-approved treatment systems, and IMO has published these. Some of these submissions contain no test results, some provide partial results or summary statistics (ranges or averages for a set of trials), and some provide full test results for the required number of trials, but not the full test data. A few manufacturers have released the test reports on their treatment systems, but most appear to consider these reports to be proprietary and confidential. Some of the agencies that review and approve treatment systems have posted or released some test reports and test data.

Boundaries and Temperatures of Arctic Marine Waters

Various researchers and multi-national programs have defined different boundaries for Arctic waters, based on such factors as topography, physical and chemical conditions, biological distributions, or political jurisdictions. These include Ekman's Arctic Region; the region addressed by the Arctic Monitoring and Assessment Program (AMAP); OSPAR's Arctic Maritime Region in the Atlantic; Spalding's Arctic Realm; the Large Marine Ecosystems defined as Arctic by UNEP or PAME; the Marine Study Units of the World Wildlife Fund's RACER Project; and the region covered by the Arctic Register of Marine Species (Ekman 1953; AMAP 1998; OSPAR 2000; Spalding 2007; Sherman & Hempel 2009; Christie & Sommerkorn 2012; PAME 2013; ARMS 2017). All of these include a core area comprising the waters north of Eurasia, North America and Greenland, and in the Atlantic, waters extending south to Baffin Bay, Davis Strait and the West Greenland Shelf, the Greenland Sea, waters north of the Iceland Shelf, and the northern and eastern Barents Sea east of the White Sea. Prior to recent modification by anthropogenic climate change, these areas had average winter surface water temperatures between around -2°C (the freezing point of seawater[6]) and 0°C, and average summer surface water temperatures below around 5°C (Sverdup *et al.* 1942; Ekman 1953).

[3] The requirements for US type approval granted by the US Coast Guard are generally similar to those for IMO type approval, but contain some significant differences, including: requiring 6 successful land-based trials and 5 successful shipboard trials; at least 1-day hold time required in land-based tests; and tests must be conducted by laboratories certified by the US Coast Guard (USEPA 2010; USCG 2012).
[4] Most bacteria in ballast water are heterotrophic, that is, they rely on other organisms or dead organic matter for food rather than synthesizing it by photosynthesis or other means.
[5] A statement regarding public release of test data was added in the 2016 guidelines, but see the discussion below in the section on "Treatment System Testing; and Access to Performance Data".
[6] The precise freezing point depends on the salinity of the water. Water with a salinity of 35 psu freezes at -1.8°C.

Each of these Arctic boundaries also include some southern waters marginal to the core region. In the Pacific region these variously include the eastern and/or western Bering Sea, with coastal waters extending to the middle or to the southern tip of the Kamchatka Peninsula; the entire or the western part of the Okhotsk Sea; the northern Kurile Islands; the Aleutian Islands; and the southern Alaskan coast from the tip of the Alaska Peninsula to near Yakutat Bay at 60° N latitude. In the Atlantic region these additional waters include the western Labrador Sea to the central or southern part of the Labrador coast or to northern Newfoundland; the East Greenland Shelf; the northeastern part of or all of the Iceland Shelf; the Faroe Plateau; Norwegian coastal waters north of 62° N latitude; the coastal waters of the Kola Peninsula; and the White Sea. These waters, sometimes termed Transitional or Subarctic, have average, pre-recent-climate change winter temperatures that are mostly below 2°C but as high as 7-8° in a few places, and average summer temperatures that are mostly below 10°C but as high as 13°C in some places (Sverdup et al. 1942; Ekman 1953).

In this paper I'll refer to the first-described waters as Arctic Core waters, and the more southern marginal or transitional waters as Subarctic waters (defined as Arctic by some studies or programs but not by others). I'll treat both Arctic Core and Subarctic waters as subregions within Arctic waters.

Source Areas and Invasion Risk

Conceptually, non-native species could arrive in Arctic marine waters from three general regions: northern hemisphere polar waters (involving introductions from one part of the Arctic to another), southern hemisphere polar waters (introductions from Antarctic/ Southern Ocean waters) and non-polar waters (introductions from warmer regions).

Northern hemisphere polar waters. Some cargo ships already transfer ballast water between different parts of the Arctic (Chan *et al.* 2012), and will likely increase with increases in commercial activity and ship traffic in the region. For some types of organisms large differences have been reported in the species present in different Arctic regions (Christiansen & Reist 2013; Josefson & Mokievsky 2013), including notable differences between Palearctic and Nearctic fauna, between fauna in the Atlantic region and elsewhere in the Arctic, and between faunas north and south of the Bering Strait (Ekman 1953). There is thus significant potential for intra-Arctic shipping to introduce species into regions where they are not native. In addition, the native biota of waters surrounding islands often lack species that are found in mainland coastal waters, creating a potential for non-native species introductions to island waters even across relatively short distances. Consistent with this, the waters around Iceland are reported to host a relatively large number of non-native species, compared to other Arctic waters (Cohen 2016).

Southern hemisphere polar waters. The marine species of the Antarctic region are almost entirely distinct from, and more richly diverse than, Arctic marine species (Ekman 1953), and thus present a substantial pool of potential invaders from habitats with similar environmental characteristics. However, the potential for transporting such organisms from the Antarctic to the Arctic in ballast water seems low. Certain types of vessels designed for work in polar regions, such as research vessels, cruise ships or fishing vessels, and possibly vessels used to explore for or extract mineral resources, might on occasion move between the two polar regions, and this could happen more often as ice retreats and commercial developments expand in the Arctic, but these types of vessels carry little ballast water.[7] Cargo ships can carry much larger quantities of ballast water, but it doesn't seem likely that cargo ship traffic, involving the

[7] These vessels may pose a greater risk of Antarctic to Arctic invasion via biofouling.

shipment of goods or raw materials, will develop between the two regions. Also, during a voyage the temperature of the water in a ship's ballast tank tends toward equilibration with ambient water temperatures,[8] so that ballast water transported from one polar region through temperate and tropical waters toward the other pole would undergo relatively rapid warming followed by cooling, which would tend to increase mortality rates for organisms carried in the ballast water. Thus, despite the existence of a pool of ecologically-suitable potential invaders, the risk of Arctic invasion via ballast water from the Antarctic is probably low.

Non-polar waters. The concept of ecological niche suggests that an introduced organism is more likely to become established in a new location if environmental conditions are similar to those of its native range. Risk assessments based on this concept are variously referred to as ecological risk modeling, bioclimate envelope modeling, or climate matching (Peterson 2003; Pearson & Dawson 2003; Bomford *et al.* 2010). For invasions into high latitude marine waters, temperature is a critical matching variable: other things being equal, organisms from source areas with water temperatures that are closer to Arctic temperatures are more likely to become established within the Arctic than organisms from warmer source areas (Gjershaug *et al.* 2009; De Rivera *et al.* 2011). Thus organisms from Boreal/Cold Temperate waters, where typical average winter temperatures are less than 10°C and typical summer temperatures range 10-17°C (Table 1), have a significant potential for establishment in the Arctic, and several studies indicate that climate change will increase that potential (Gjershaug *et al.* 2009; De Rivera *et al.* 2011; Ware *et al.* 2014, 2016).[9] In addition, the available data suggest that most of the ballast water currently discharged in the Arctic comes from Boreal/Cold Temperate regions.[10] Besides being the source of less Arctic-discharged ballast water, organisms from Warm Temperate regions seem less likely, and Tropical/Subtropical organisms very unlikely, to establish.[11]

Table 1. Typical average surface water temperatures in northern hemisphere and tropical/subtropical biogeographic regions. Arctic Core and Subarctic regions are as described in the text; the other region boundaries are derived from Ekman (1953), with temperatures from Sverdrup *et al.* (1942).

Region	Winter	Summer
Arctic Core	-2° to 0°	≤5°
Subarctic	mostly ≤2°, sometimes to 8°	mainly 5-10°, sometimes to 13°
Boreal/Cold Temperate	-1° to mostly <10°, sometimes to 15°	mostly 10-17°, sometimes to 25°
Warm Temperate	10-18°	16-23°, sometimes to 27°
Tropical/Subtropical	mostly 20-28°, sometimes ≥15° or ≤33°	

Treatment Modes

[8] Changes in ballast water temperatures of up to 16°C have been recorded on voyages to the Arctic (Chan *et al.* 2014).

[9] There has already been a rise in average summer water temperatures of around 2°C since the early 1980s in some parts of the Arctic (Timmermans *et al.* 2017).

[10] Most of the ballast water discharged in the Canadian Arctic (Chan *et al.* 2012, Fig. 12 total discharge volumes; all of the NW Atlantic region and much of the NE Atlantic Region are Boreal/Cold Temperate) and nearly all of the ballast water discharged at Svalbard (Ware *et al.* 2014, Fig. 3) is from Boreal/Cold Temperate locations. On the other hand, it appears that a substantial portion but less than half of the ballast water discharged at Red Dog, Alaska in the Chukchi Sea is from Boreal/Cold Temperate locations (Verna 2014, Fig. 1.7; note also Verna *et al.* 2016, Fig. 3: most of the ballast water discharge in Alaska appears to be from Boreal/Cold Temperate locations).

[11] These statements refer to organisms found in surface waters. Even in the tropics, the deepest water may have temperatures similar to Arctic surface waters, and some organisms occur naturally in both tropical abyssal waters and Arctic surface waters (Ekman 1953). However, since ballast water is pumped from near the ocean surface, deep-water organisms are unlikely to be loaded into ballast tanks.

Different shipboard ballast water treatment systems *operate* and *take effect* during different phases of the ballasting/deballasting cycle. Since both ambient conditions and ballast water temperatures can vary over a voyage, assessments of treatment system performance relative to these conditions must take into account the points during the voyage when the system operates and has its impact. Concerns include both the proper operation of the system components (for example, do the filters, chemical injectors, UV reactors, etc., physically function as intended in the water temperatures and ambient conditions encountered?) and with their effective action (for example, do the injected biocides achieve the necessary rate of kill at the water temperatures encountered?). The land-based and shipboard trials required for type approval are intended to test both system operation and effective action.

Consider shipboard treatment systems that include a filtration step. These typically conduct filtration during ballasting, so that the untreated backwash water can be discharged while the ship is still in the source region. Thus, for ballast water loaded in temperate waters and discharged in the Arctic, the relevant concern is for the proper operation and effective action of the filters under temperate conditions, not under Arctic conditions. Only for ballast water loaded in the Arctic is filter operation and action under Arctic conditions relevant.

Most treatment systems conduct some or all of their operations during ballasting. These include, among others: the particle separation processes of filtration and hydrocyclonic treatment; injection of biocides; *in situ* creation of oxidizing biocides through electrolysis; and exposure to UV radiation. Very few systems apply treatments during the voyage, such as directly injecting biocides into the ballast tanks, or circulating water out of the ballast tanks, through a treatment unit, and back into the tanks. In other cases, processes that occur in the tank over time during the voyage affect the treatment outcome, including: the killing or inactivation of organisms from prolonged exposure to a biocide delivered during ballasting; organism recovery, repair and re-growth; and the breakdown of biocides to concentrations that can safely and legally be discharged. Operations that are applied during deballasting include, for most UV systems, a second exposure to UV radiation during discharge; for some biocide-using systems, injection of an agent to neutralize the biocide; and for deoxygenation systems, reoxygenation of the ballast water.

The Analysis

I compiled publicly available test data for shipboard ballast water treatment systems that were type-approved under IMO guidelines, and analyzed these data to address the following questions:

(1) How much data are available for assessment of system performance in cold water?

(2) Are there trends in the data showing better or poorer performance at lower temperatures?

(3) What range of performance is exhibited in these data by individual treatment systems and types of treatment systems at lower temperatures?

I also reviewed temperature limits for treatment system operation disclosed by treatment system manufacturers or by agencies that assessed treatment systems for type approvals or other approvals.

Data Acquisition

I obtained test reports, test data and certification letters for shipboard treatment systems from: internet searches, including IMO, US Coast Guard and US Environmental Protection Agency websites and websites of government agencies that have granted IMO type approvals; inquiries of treatment system manufacturers and testing laboratories; Freedom of Information Act requests submitted to the US Coast Guard; and a Public Records Act request submitted to the California State Lands Commission. I obtained information on detection limits from the test laboratories, or determined detection limits from the test reports and associated documents (see Cohen *et al.* 2017 for details). Treatment systems included in the database are listed in Appendix B, with citations for the test reports, QA/QC documentation, certifications, and other relevant documents.

I considered the various models of a treatment system that differ only in the "Treatment Rated Capacity" (TRC, the maximum flow the treatment unit is designed to handle), or models with different types or brands of filter but with the same or similar nominal pore size, or models that are or are not explosion-proof, to be merely different versions of the same treatment system. I considered versions of a treatment system made by the same manufacturer but that use different processes or different forms of a biocide, or that have different environmental requirements, or that otherwise appear to differ significantly based on the manufacturer's description or on the separate testing required for type approval, to be different treatment systems. Some other authorities have classified a few treatment systems differently and thereby produced counts of the total number of approved treatment systems that are slightly different from mine.

Performance Assessment

In each trial, I assessed treatment system performance in terms of the degree to which the concentration of viable organisms in the treated discharge was reduced relative to the concentration in the discharge from the control stream. I included trials in the analysis only if data were available on the temperature of the intake water and on organism concentrations in both treated and control discharge. If the mean value of a variable was provided for a set of n trials instead of individual values for each trial, I analyzed these as n trials using the mean as the value for each trial. If only the range of values for a set of n trials was provided, I analyzed these as n trials using the midpoint of the range as the value for each trial.

I calculated performance, or degree of reduction, as the viable organism concentration in the control discharge divided by the viable organism concentration in the treated discharge. When a concentration was reported as "not detected" or below a detection limit, I used the detection limit as the concentration. When a concentration was reported as above a given number, *i.e.* ">x", I used x as the concentration. In these cases the calculated degree of reduction was not the actual reduction achieved in the trial, but rather the minimum reduction that can be calculated from the data, the actual reduction being indeterminable. Thus, in these cases the degree of reduction used in the analysis understates the actual reduction. I excluded from the analysis trials in which control and treated discharge concentrations were both reported as not detected, or both reported as above some number, or either was reported as TNTC or "too numerous to count." Mean performance over a set of trials was calculated as the sum of the concentrations in the control discharges divided by the sum of the concentrations in the treated discharges.

I conducted sensitivity analyses by analyzing certain subsets of the initial data sets to see whether the subsets yield results similar to results from the initial data set. I reference these analyses in the text and provide the results in Appendix A.

Phytoplankton group. IMO's discharge standard limits the concentrations of "viable" organisms in discharges while US regulations limit the concentrations of "living" organisms. In tests of ballast water treatment systems, viable phytoplankton are counted by observing growth in an appropriate medium after serial dilutions, often referred to as the Most Probable Number or MPN method; living phytoplankton are counted using stains to distinguish live from dead cells. In this analysis, I used viable organism counts if available; if not, I used living organism counts; and if these were not available, I used counts of chlorophyll-containing cells (based on chlorophyll autofluorescence) or counts of intact cells, which include both live and dead cells. In some trials, control discharge was analyzed by counting intact cells but treated discharge was analyzed using stains to distinguish live from dead cells, presumably on the assumption that most intact cells in untreated (control) discharge are viable, but not in treated discharge. In these trials I calculated performance using the data provided, which likely overestimated the degree of reduction.

The "phytoplankton group", consisting of all organisms between 10 and 50 μm in minimum dimension, is mostly actual phytoplankton, but also includes a small number of zooplankton within this size range, termed "microzooplankton." In trials where both phytoplankton and microzooplankton concentrations were assessed, I added the two together to determine control discharge and treated discharge concentrations for the "phytoplankton group" before calculating the degree of reduction.

In some cases, laboratories concluded that most of the living phytoplankton cells or microzooplankton counted in the treated discharge were non-viable or dead, based on further tests that measured photosynthetic efficiency or that incubated samples to assess delayed mortality, cell degradation or reproductive potential. Since these assessments did not quantify the concentration of viable phytoplankton, I calculated performance using the quantitative results from the cell counts, which likely underestimated the degree of reduction.

Bacteria. Five types of bacterial assessments of treated discharge were conducted in different trials. Three of these, which target the three indicator bacteria limited by the IMO discharge standard, were conducted in most trials. However, in nearly all of these trials the control discharge concentrations were near or below the detection limits, so that the resulting data were either of little value or completely useless for determining treatment system performance (Cohen & Dobbs 2015). Some trials counted intact bacteria cells which, as in the counts of intact phytoplankton cells, include both live and dead cells. Since the proportion of dead cells is likely to be higher in treated than in control discharge, using total bacteria counts to assess performance could systematically underestimate the degree of reduction. Finally, many of the trials conducted assessments of culturable heterotrophic bacteria (CHBs), which count the number of bacteria that develop into colonies on a standardized growth medium under specified incubation conditions. CHBs comprise only a small fraction of the bacterial cells present in environmental samples, typically less than one percent and sometimes less than one-tenth of one percent. Nevertheless, I used the CHB data for the analysis because the absence of systematic bias makes them the least problematic of the available bacteria data.

Results

Zooplankton

Data needed for analysis of zooplankton treatment (trials providing data on intake water temperature as well as organism concentrations in treated and control discharge) were acquired for a total of 985 trials. These include at least one trial for each of 67 treatment systems type-approved under IMO guidelines, representing 93% of the 72 IMO type-approved systems. These include data for 60 (97%) of the 62 treatment systems that have been accepted as Alternate Management Systems (AMS) by the US Coast Guard,[12] and for all 6 of the US type-approved treatment systems (Table 2).

Table 2. Number of trials and treatment systems analyzed.

Organism group	IMO Type-approved		AMS		US Type Approved	
	Trials	Treatment Systems	Trials	Treatment Systems	Trials	Treatment Systems
Zooplankton	985	67	909	60	82	6
Phytoplankton	976	66	902	60	82	6
Bacteria	539	46	494	43	40	4

The intake water temperatures in these trials ranged from 0.7-38°C, and the performance ranged from a 5-fold reduction in zooplankton concentration in the least impressive trial to a 1.6-million-fold reduction in zooplankton concentration in the most impressive trial. Figure 1 is a plot of performance versus temperature in these trials, which reveals no apparent tendency toward better or poorer performance at lower temperatures. Plots of subsets of these data—categorized by major particle separation and disinfection methods, by salinity, by holding time, and by land-based or shipboard tests—in Appendix A also show no significant trend.

Figure 2 shows how the 985 trials analyzed—that is, the test data on performance and temperature that are available to the public—are distributed among different ranges of intake temperatures. There are relatively few data below 10°C (typical temperatures for the Subarctic surface waters, especially in the summer), very few below 5°C (typical temperatures for the Arctic Core in the summer), hardly any below 2°C (common in the Subarctic in the winter), and none between 0.7°C and the freezing point of seawater just above -2°C (Arctic Core winter water temperatures). Figures A-16 and A-17 in Appendix A show similar distribution patterns when restricted to trials of AMS systems and US type-approved systems.

[12] Treatment systems accepted as AMS can be used in US waters for up to 5 years instead of a US type-approved system (USCG 2012). This was intended to serve as an interim program until sufficient treatment systems were type approved.

Figure 1. Treatment system performance vs. water temperature. System performance calculated as the zooplankton concentration in the control discharge divided by the zooplankton concentration in the treated discharge. y=656x+50850, n=985, r2=0.001, p<0.325.

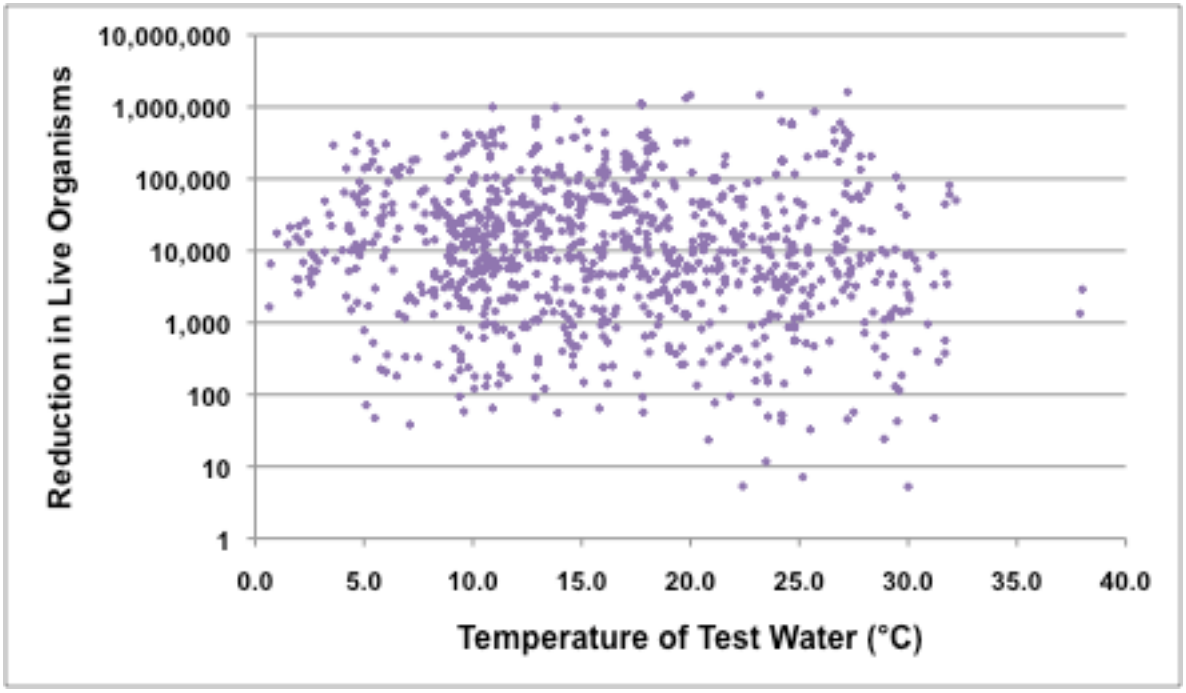


Figure 2a. Zooplankton data: Number of trials in the overall data set per temperature bracket, for IMO type-approved treatment systems.

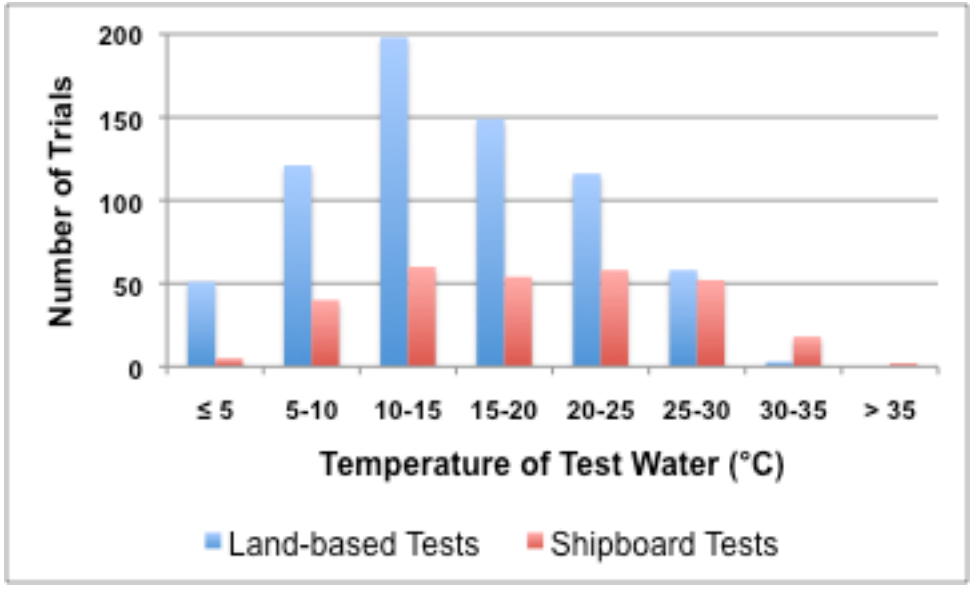


Figure 2b. Zooplankton data: Number of trials per temperature bracket, for IMO type-approved treatment systems, for temperatures of 0-5°C.

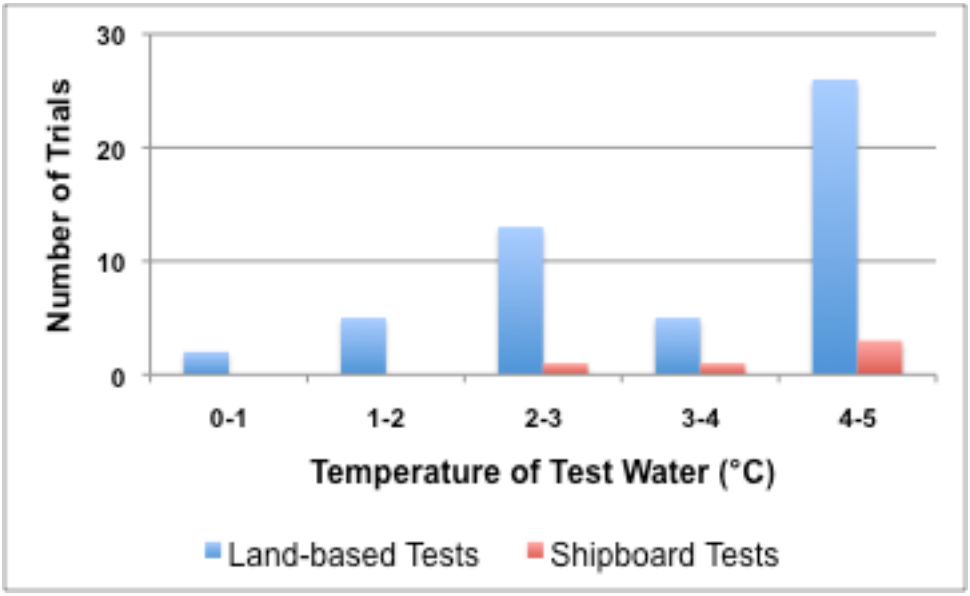


Figure 3 shows, by temperature brackets, the number of treatment systems for which there is at least one trial with publicly available data on system performance and temperature. For temperatures below 5°C, there are only 15 treatment systems with such data.

Figure 3. Zooplankton data: Number, per temperature bracket, of IMO type-approved treatment systems with at least one trial in the overall data set.

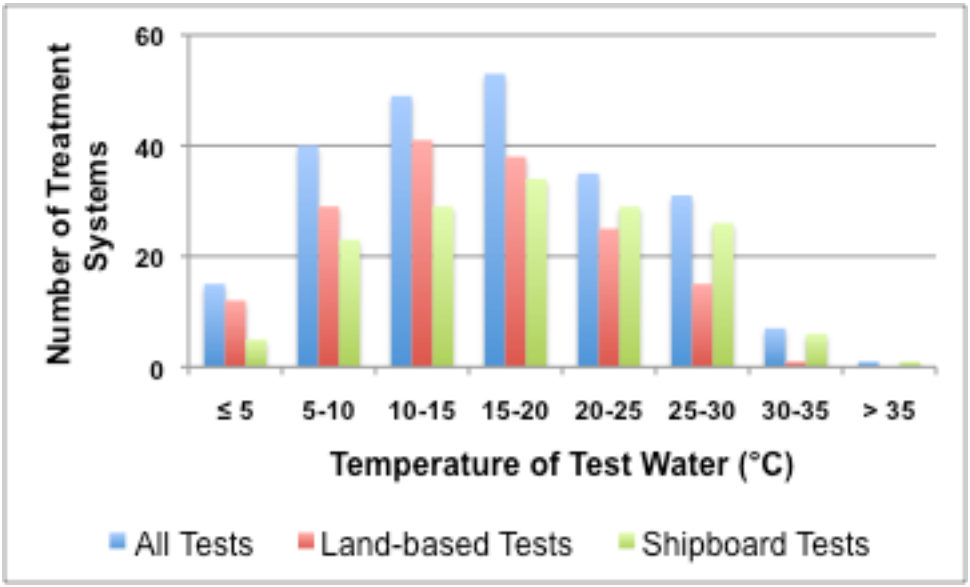


Figure 4 again shows the number of trials per temperature bracket, but sorted according to the laboratories that conducted the trials. Three laboratories (out of the 19 represented in the database) conducted nearly all of the land-based trials with intake water temperatures below 5°C as well as all of the shipboard trials in that temperature bracket. These laboratories were: the Norwegian Institute for Water Research (NIVA), whose land-based test facility is located at Solbergstrand, near Oslo, Norway; DHI, with a test facility at Hundested, Denmark; and the Center of Marine Environmental Measurements (CMEM), at First Institute of Oceanography, State Oceanic Administration, which has conducted land-based tests at several sites, including Dalian, Weihai and Qingdao in northern China and in Zhejiang in East China. Other ballast water test facilities apparently don't have access to seawater this cold.

Figure 4a. Zooplankton data: Number of land-based trials in the overall data set per temperature bracket, by laboratory, for IMO type-approved treatment systems.

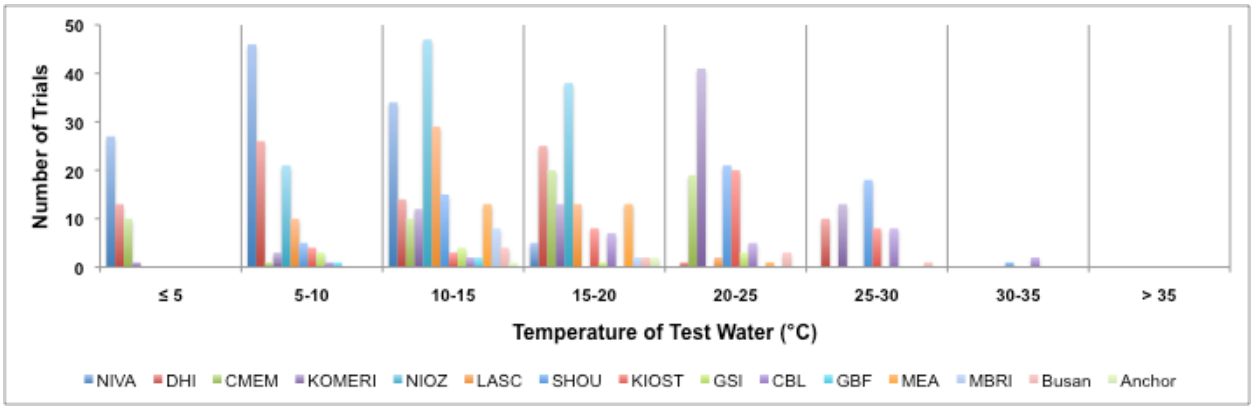
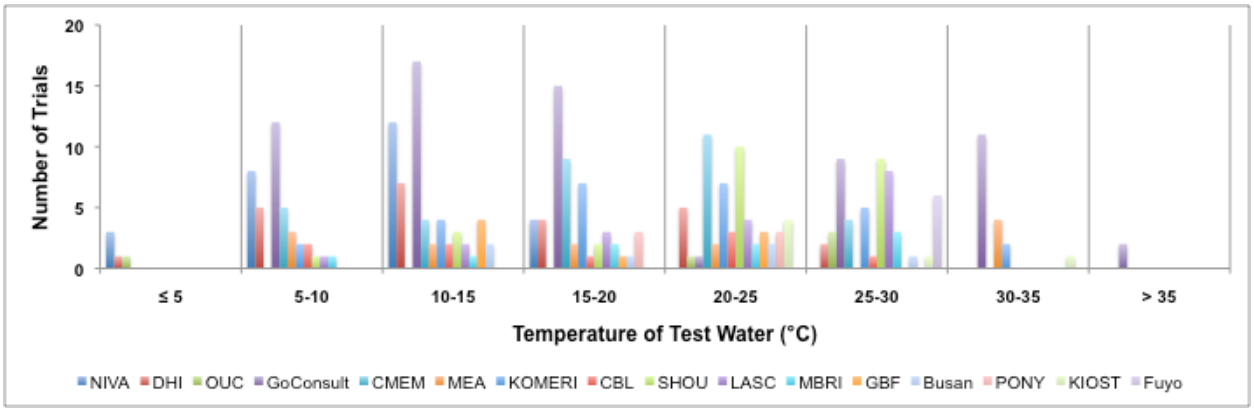


Figure 4b. Zooplankton data: Number of shipboard trials in the overall data set per temperature bracket, by laboratory, for IMO type-approved treatment systems.



Figures 5 to 8 compare different treatment systems by their average performance and by their poorest performance. I graphed the results for all treatment systems with 5 or more trials in the entire data set, and for the three coldest temperature brackets (0-5°C, 5-10°C and 10-15°C) graphed treatment systems with 3 or more trials in the data for the bracket. Treatment systems were sorted into one of four general treatment types:

- Biocide, including both injected biocides and biocides generated in situ by electrolytic processes;
- UV;
- UV+, which is either catalytically enhanced UV process or UV combined with another treatment process (not including particle separation); or
- Other, including deoxygenation, heat treatment, and membrane filtration.

Figure 5. Average and smallest zooplankton reductions by treatment system and treatment type, for all trials. Treatment systems with ≥5 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

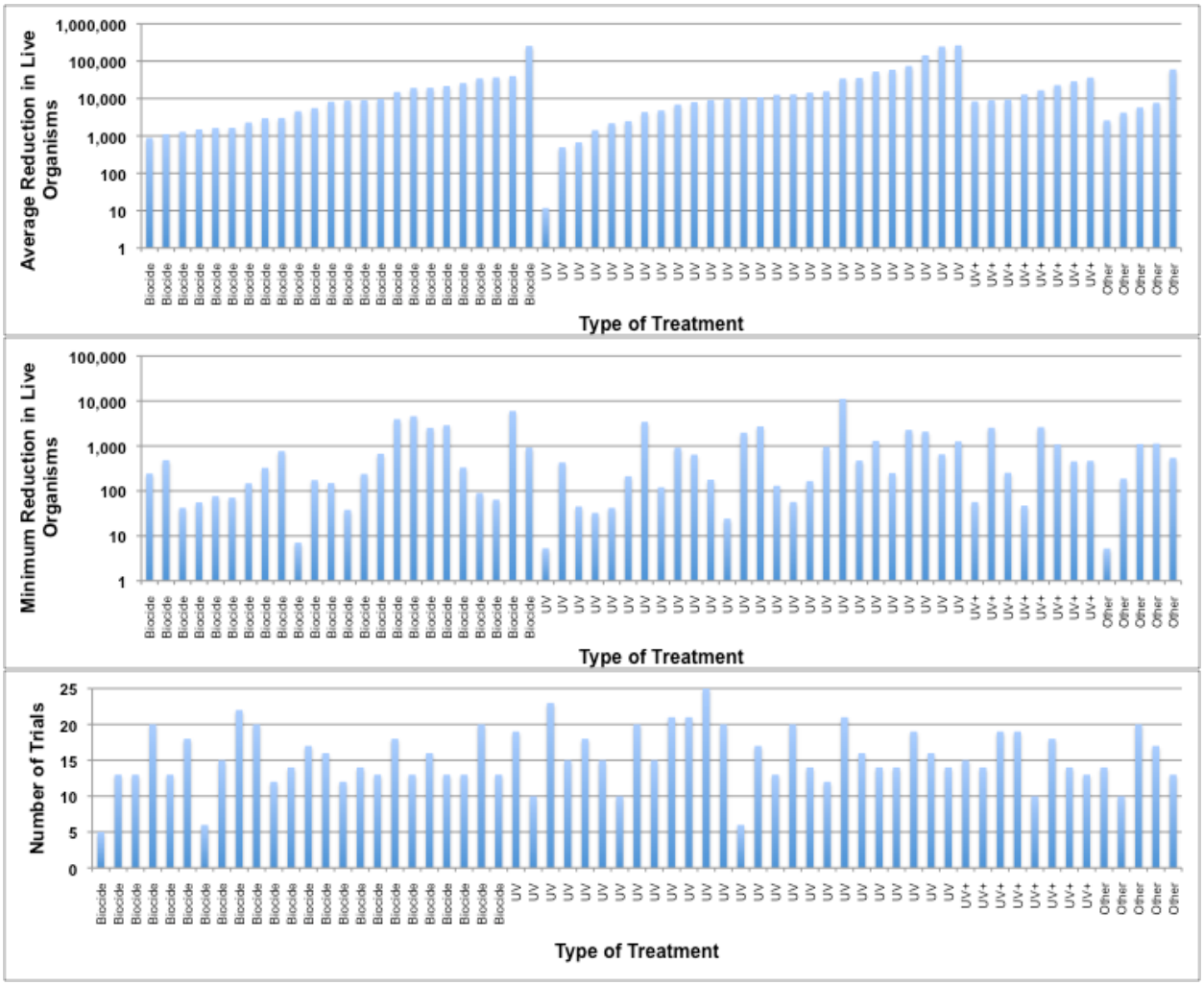


Figure 6. Average and smallest zooplankton reductions by treatment system and treatment type, for trials with intake water temperatures between 0° and 5°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

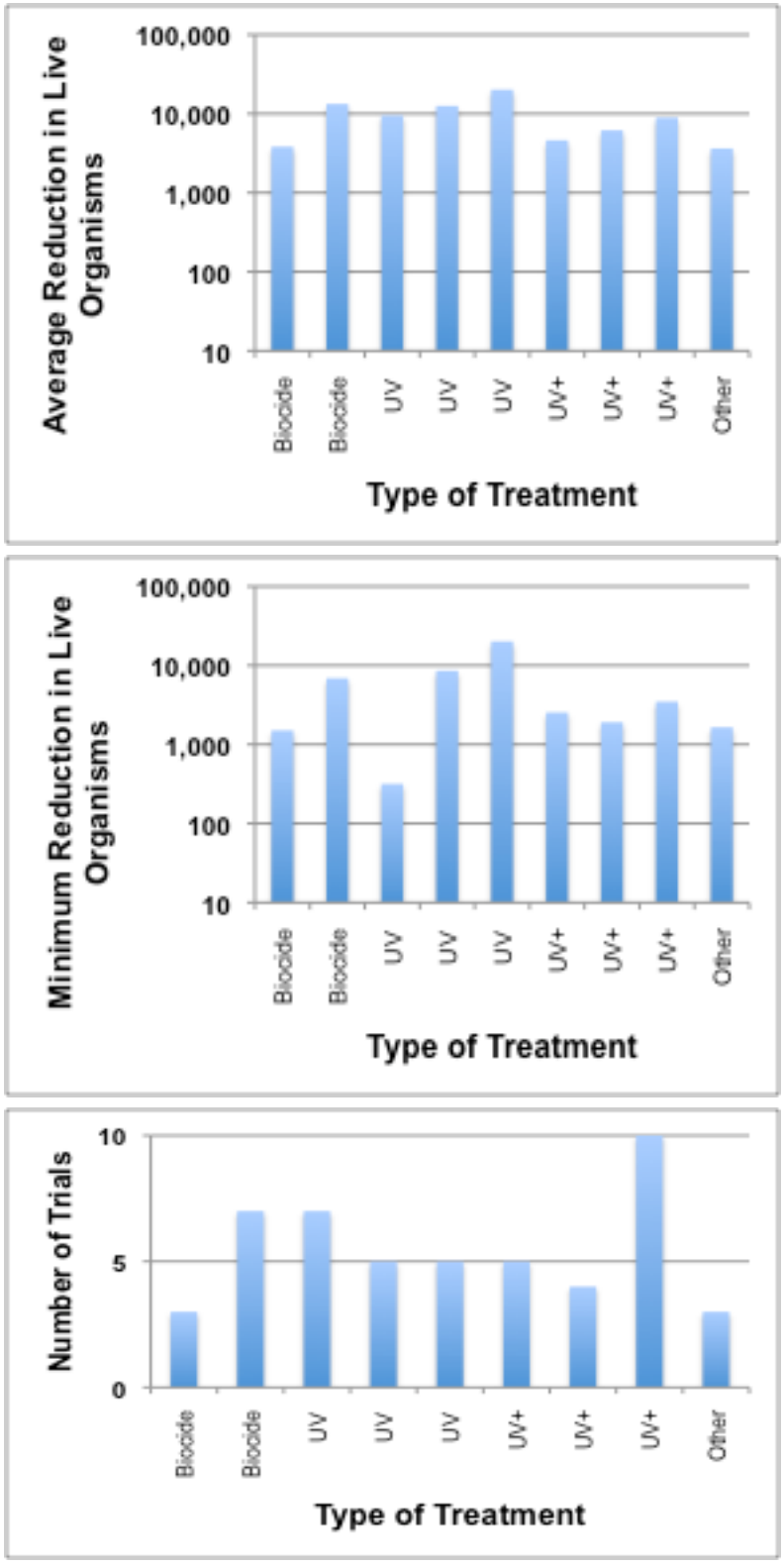


Figure 7. Average and smallest zooplankton reductions by treatment system and treatment type, for trials with intake water temperatures between 5° and 10°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

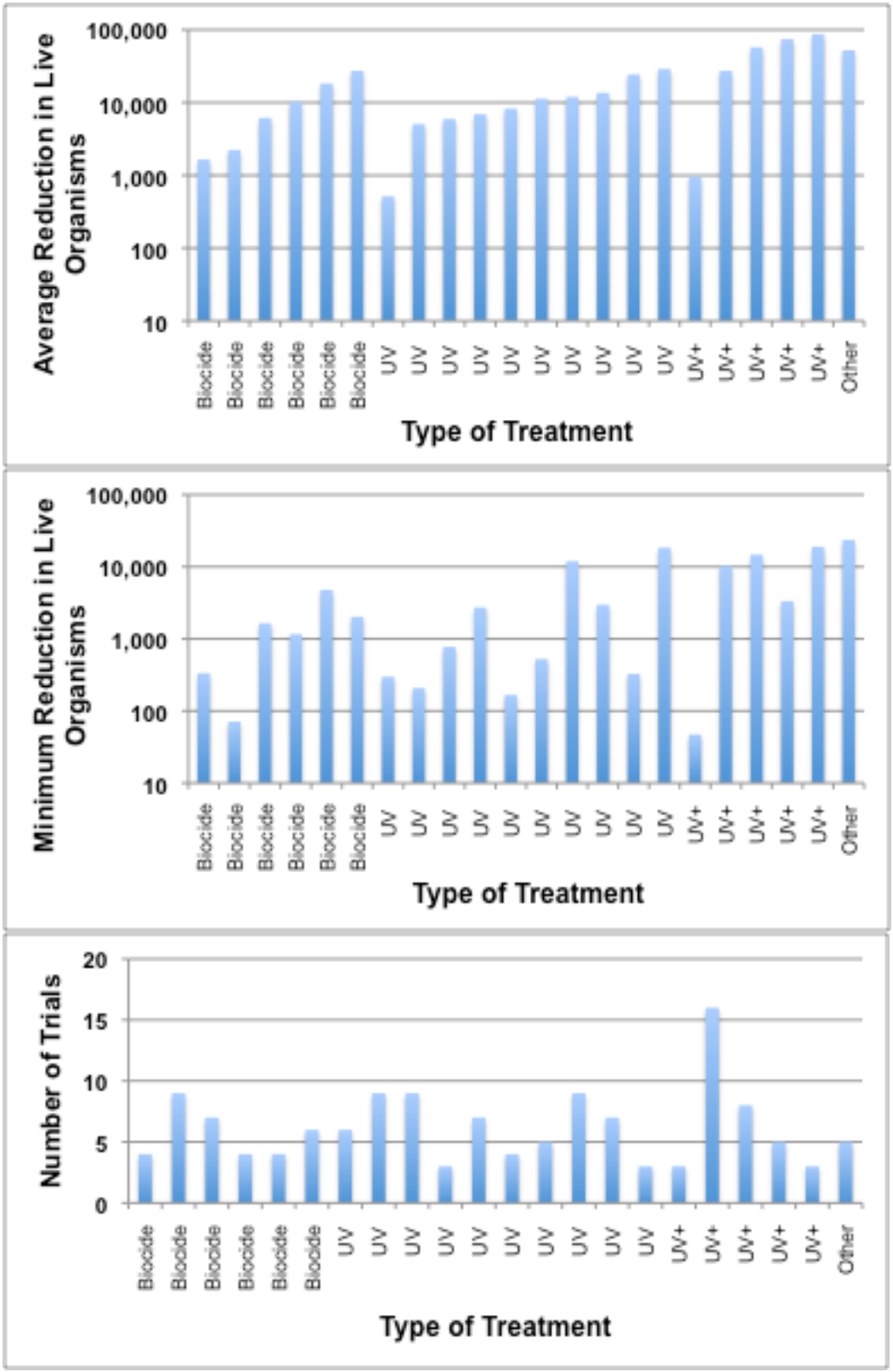
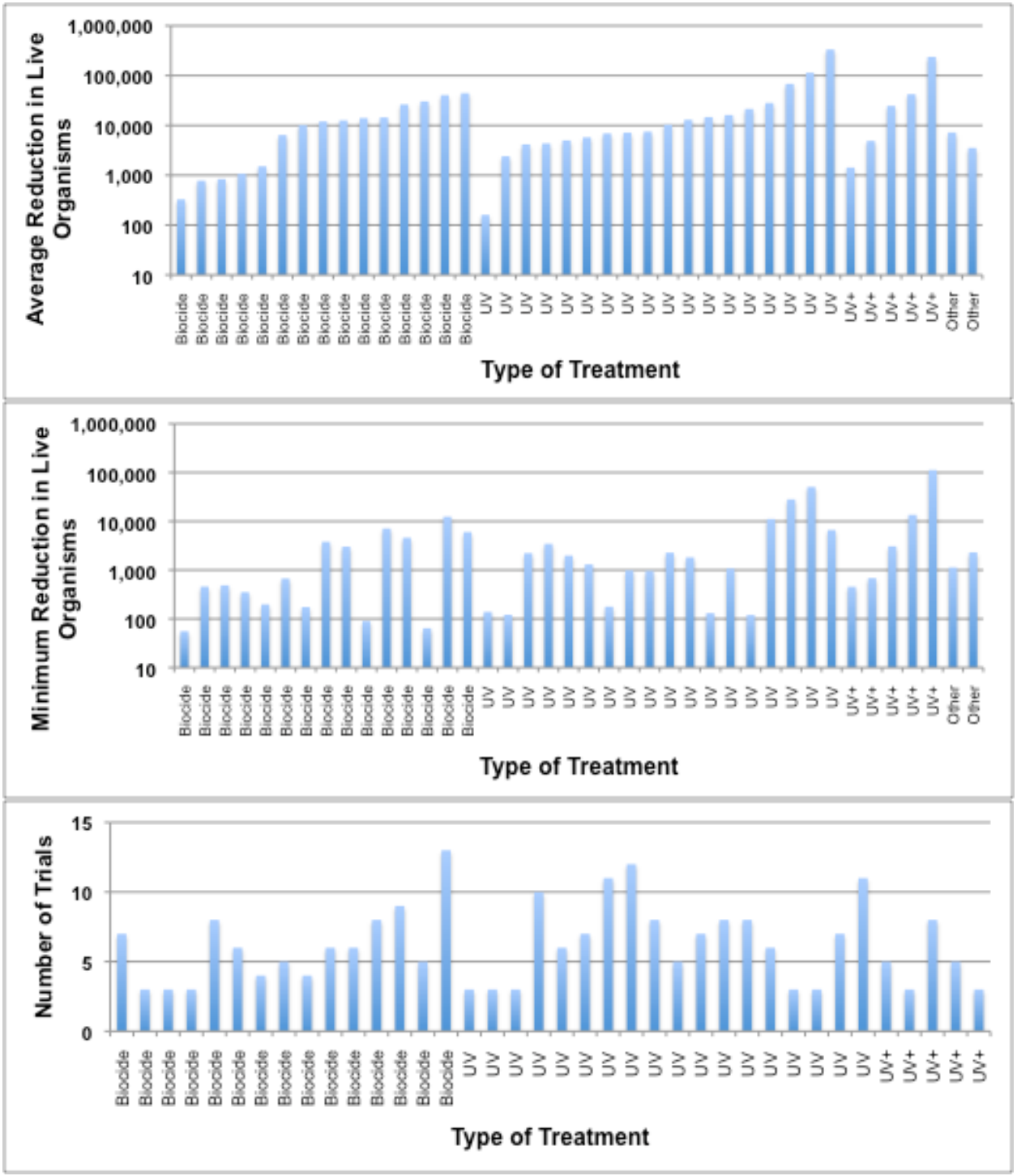


Figure 8. Average and smallest zooplankton reductions by treatment system and treatment type, for trials with intake water temperatures between 10° and 15°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.



No clear pattern emerges from these graphs in terms of one major type of treatment consistently performing better than another major type, or in terms of trends in performance over the three cold temperature brackets. What does emerge is the great range in performance, both overall and within major treatment types. For example in Figure 5, with graphs based on the entire data set, the average performance of 29 Biocide-based treatment systems ranges over more than 2 orders of magnitude, the performance of 26 UV systems over more than 4 orders of magnitude, the performance of 8 UV+ systems over approximately one-half of an order of magnitude, and the performance of 5 "Other" (mainly deoxygenation-based) systems, about one-and-a-half orders of magnitude. Large variations in performance can also be seen in trials within temperature bands, and in performance assessed in terms of the smallest rather than the average reduction.

Phytoplankton

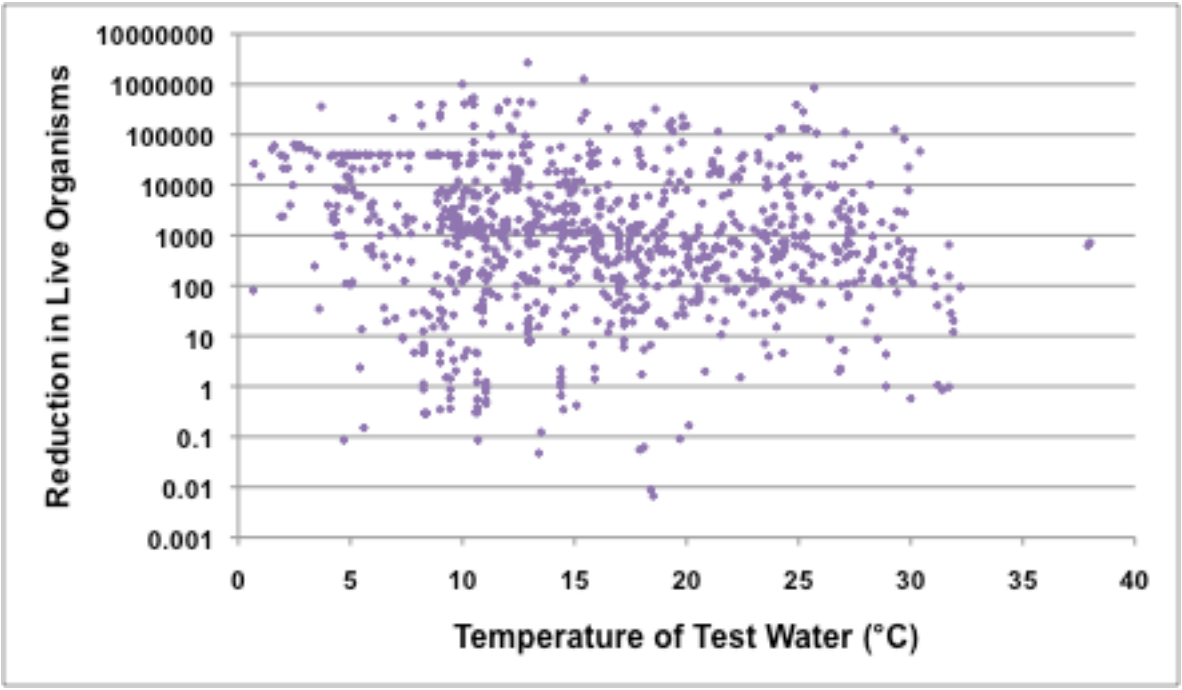
Data for an analysis of phytoplankton treatment were acquired for a total of 976 trials. These include at least one trial for each of 66 treatment systems type-approved under IMO guidelines, for each of 60 AMS, and for all 6 of the US type-approved treatment systems (Table 2). The numbers of trials and treatment systems are the same as or slightly less than the numbers for analysis of zooplankton. With few exceptions, if a trial yielded performance data on zooplankton it also yielded performance data on phytoplankton, and vice versa.

The range of intake water temperatures is the same as for the zooplankton trials (0.7-38°C), but the range of performance is considerably greater. In 33 trials (3% of the phytoplankton trials), the treated discharge had higher concentrations of phytoplankton than the untreated (control) discharge—in one case, 150 times higher. At the other extreme the largest reduction in phytoplankton concentrations, a 2.7-million-fold reduction, is larger than the largest reduction in zooplankton concentrations.

As with zooplankton, plots of performance versus temperature show no trend toward better or poorer performance at lower temperatures. The plot for all 976 trials is in Figure 9; the plots of subsets of these trials also show no significant trends.

Because the data set for phytoplankton analysis is based on nearly the same trials as the data set for zooplankton analysis, graphs showing the distribution of trials, treatment systems and laboratories across the temperature brackets will be nearly indistinguishable from those for zooplankton (Figures 2 to 4), and so are not included here.

Figure 9. Treatment system performance vs. water temperature. System performance calculated as the phytoplankton concentration in the control discharge divided by the phytoplankton concentration in the treated discharge. y=-767x+36164, n=976, r2=0.002, p<0.140.



Figures 10 to 13 graph data on the performance of individual treatment and treatment types, in different temperature brackets. These figures correspond to Figures 5 to 8 for zooplankton, and were prepared in the same way. Again, as with zooplankton, no major type of treatment appears overall to be superior to the others, and no trend in performance relative to temperature is obvious. What can be seen is a range of performance even greater than that seen for zooplankton. In Figure 10, the average performance of 23 Biocide-based treatment systems ranges over more than 3 orders of magnitude, the performance of 26 UV systems over 5 orders of magnitude, the performance of 8 UV+ systems over nearly 2 orders of magnitude, and the performance of 4 "Other" treatment systems over nearly 4 orders of magnitude.

Figure 10. Average and smallest phytoplankton reductions by treatment system and treatment type, for all trials. Treatment systems with ≥5 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

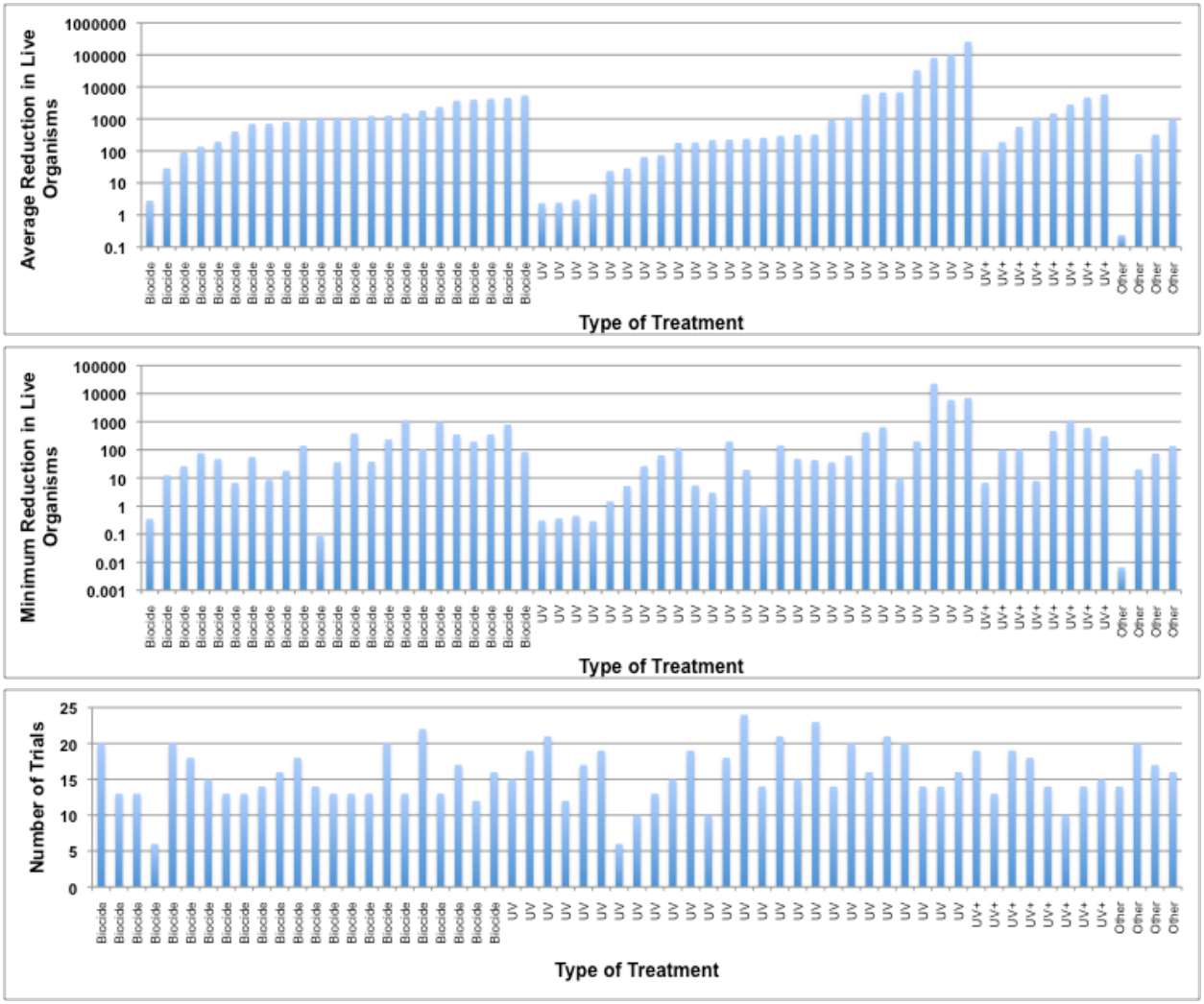


Figure 11. Average and smallest phytoplankton reductions by treatment system and treatment type, for trials with intake water temperatures between 0° and 5°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

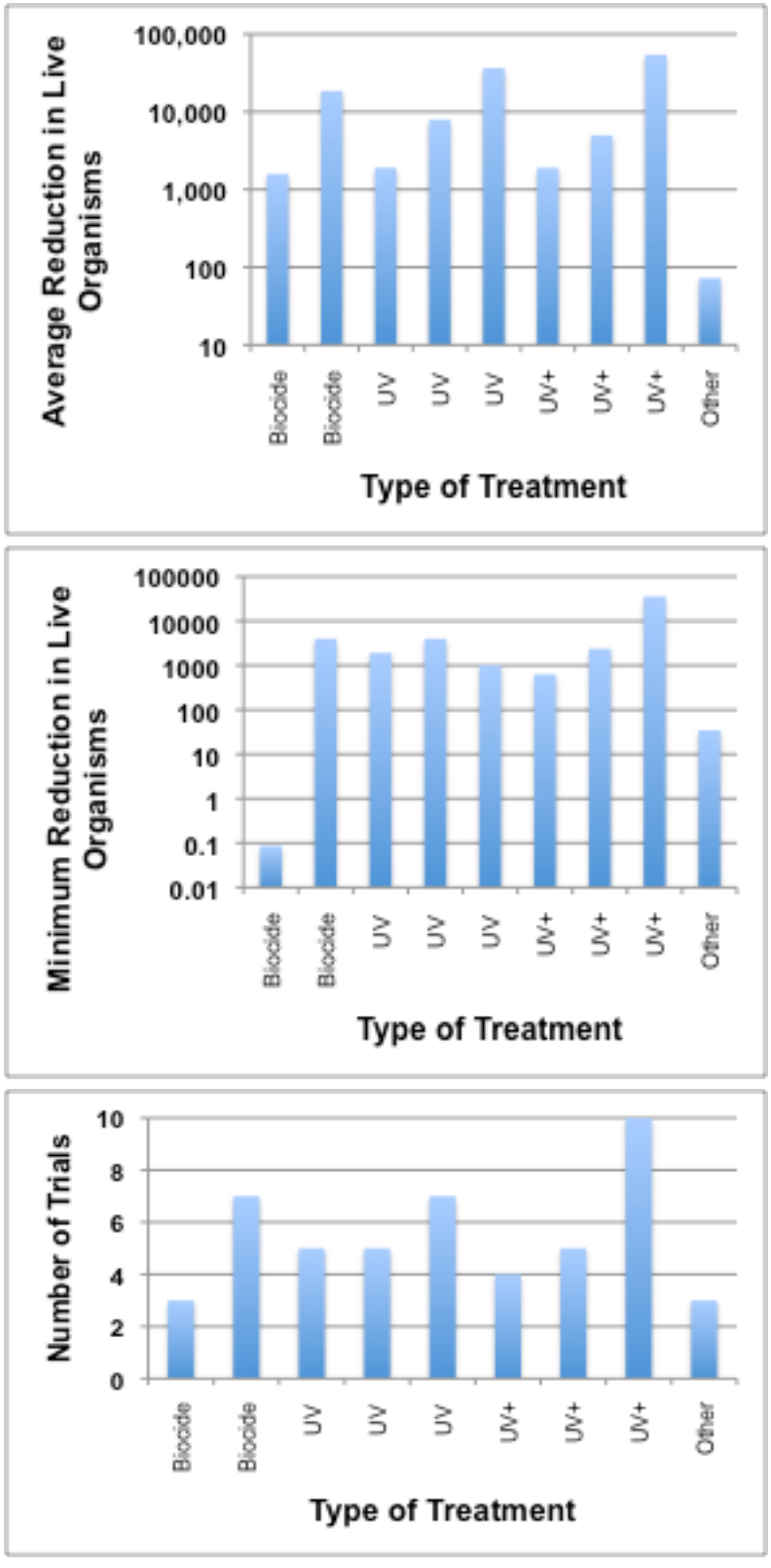


Figure 12. Average and smallest phytoplankton reductions by treatment system and treatment type, for trials with intake water temperatures between 5° and 10°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

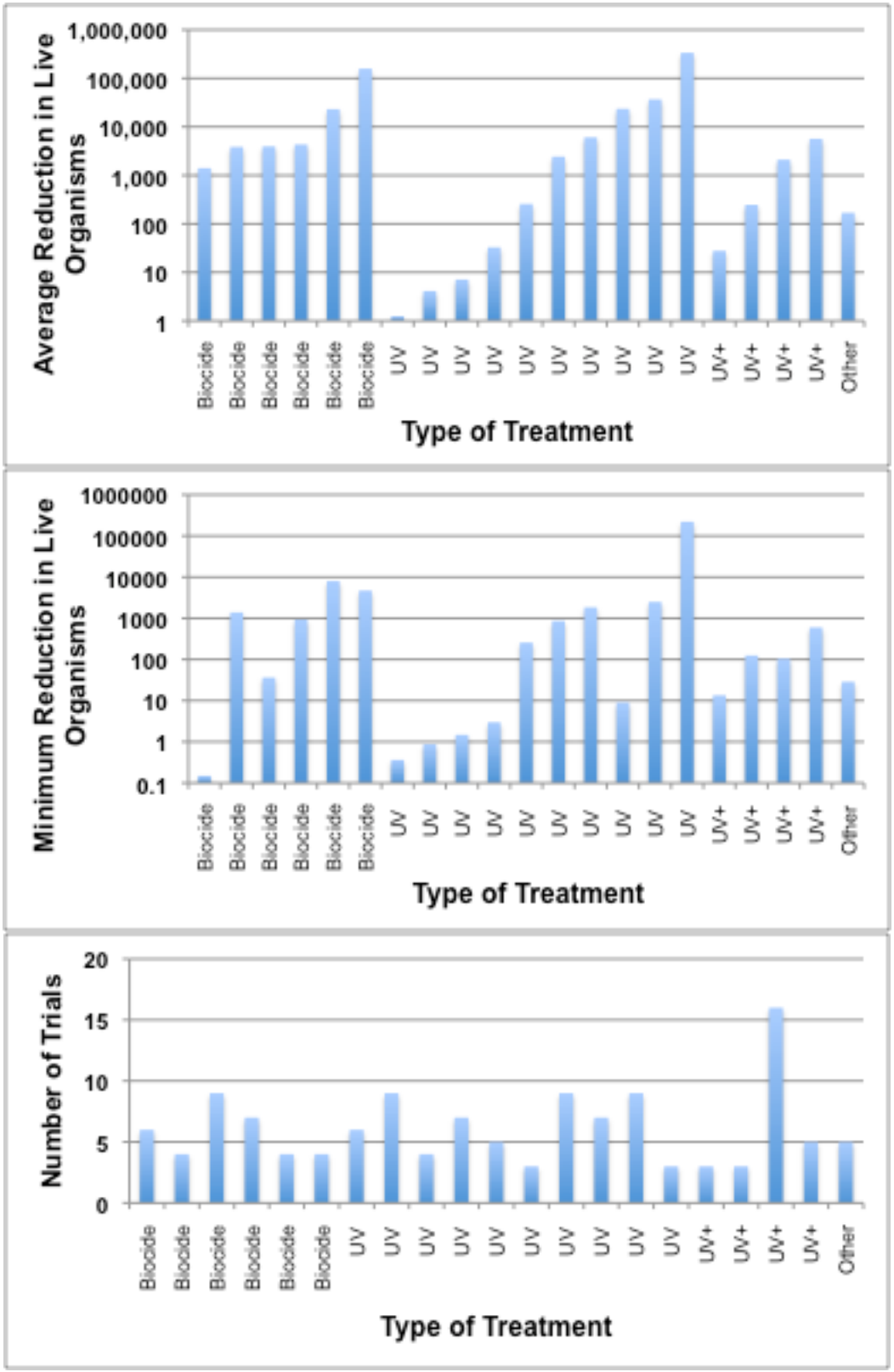
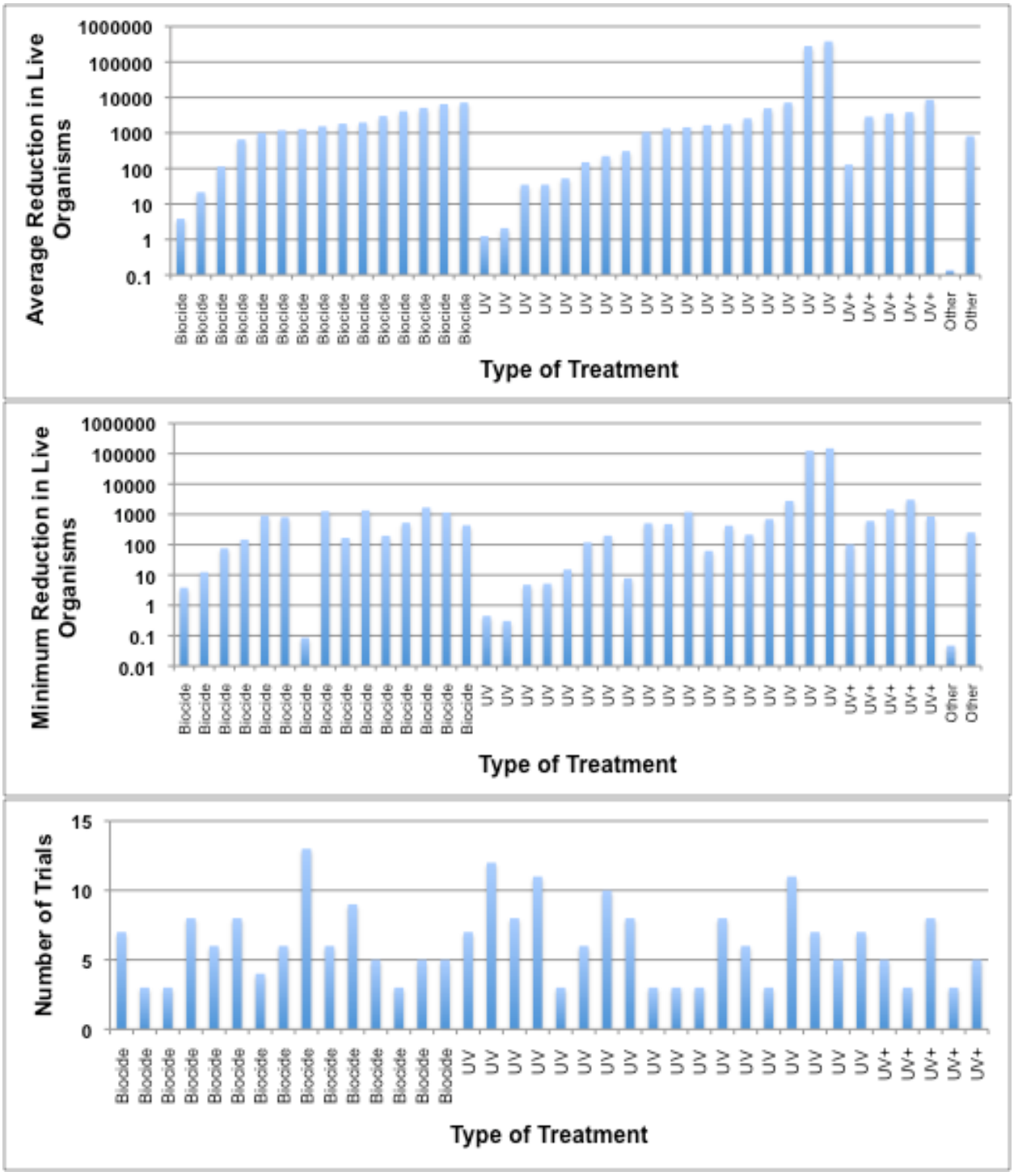


Figure 13. Average and smallest phytoplankton reductions by treatment system and treatment type, for trials with intake water temperatures between 10° and 15°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.



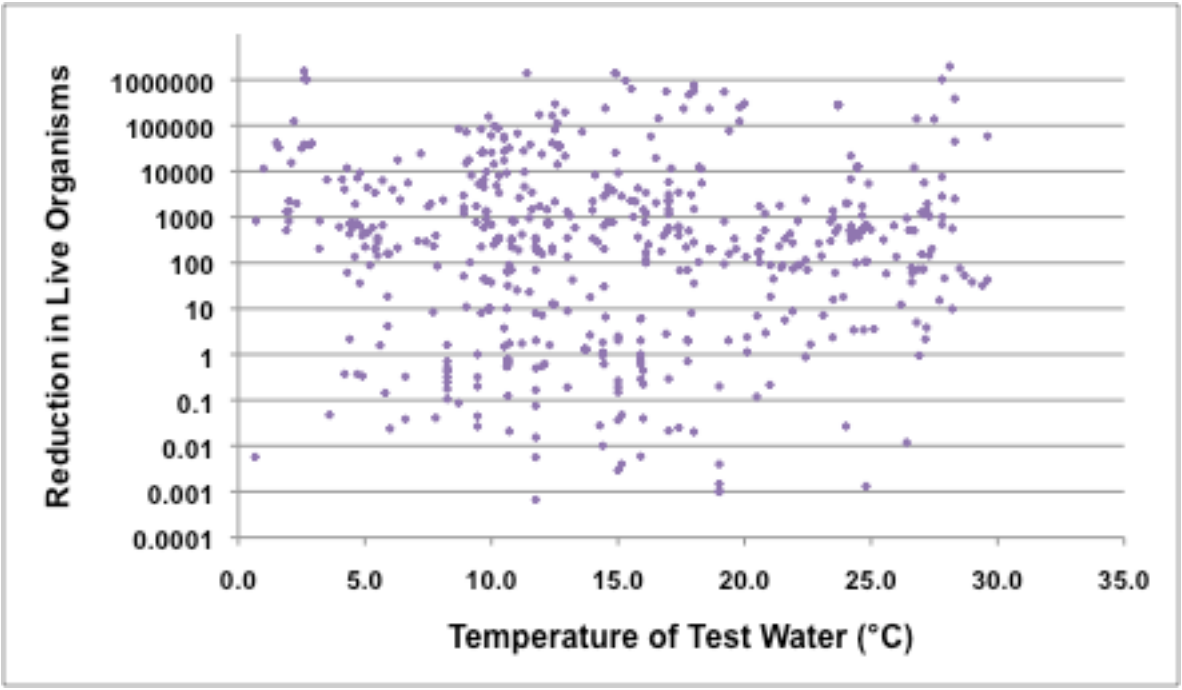
Bacteria

Data for an analysis of the treatment of culturable heterotrophic bacteria (CHB) were acquired for a total of 539 trials. These include at least one trial for 46 of the 72 treatment systems type-approved under IMO guidelines, for 43 of the 60 AMS, and for 4 of the 6 US type-approved treatment systems (Table 2).

The range of intake water temperatures is narrower than for the zooplankton and phytoplankton trials analyzed (0.7-29.6°C), but the range of performance is greater. In 74 trials (14% of the CHB trials), the treated discharge had higher concentrations of CHBs than the untreated (control) discharge—in one case, 1,500 times higher. At the other extreme, the largest reduction in the CHB data is a 2-million-fold reduction, greater than the largest reduction in zooplankton concentrations but smaller than the largest phytoplankton reduction.

Similar to both zooplankton and phytoplankton, plots of performance versus temperature show no trend toward better or poorer performance at lower temperatures. The plot for all 539 trials is in Figure 14; the plots of subsets of these trials also show no significant trends.

Figure 14. Treatment system performance vs. water temperature. System performance calculated as the CHB concentration in the control discharge divided by the CHB concentration in the treated discharge. y=237x+37807, n=539, r2=0.0001, p<0.833.



The relative distribution of trials, treatment systems and laboratories across the different temperature brackets is generally similar to that for zooplankton, with relatively few data below 10°C and 5°C, and hardly any below 2°C (Figures 15 & 16).

Figure 15a. CHB data: Number of trials in the overall data set per temperature bracket, for IMO type-approved treatment systems.

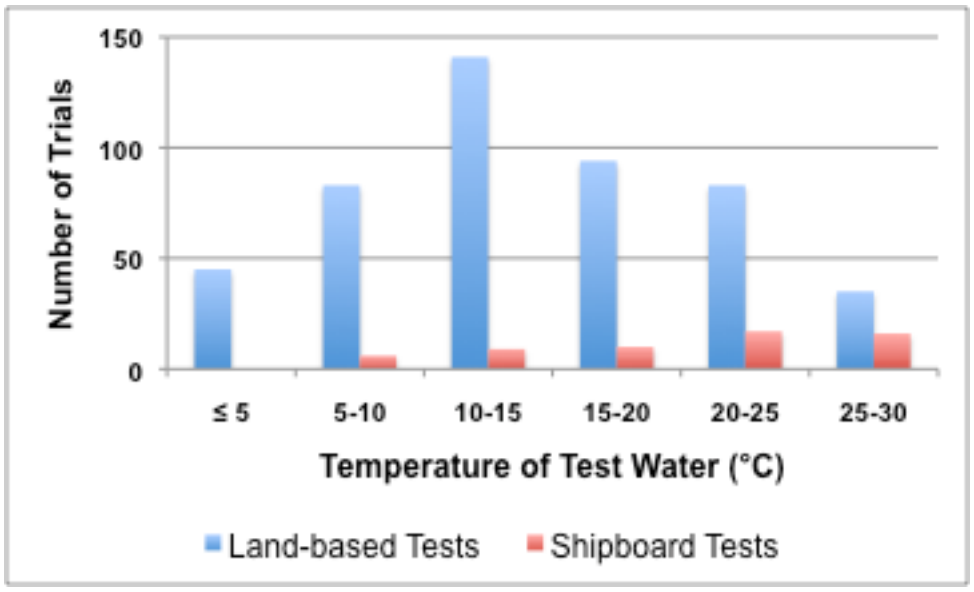


Figure 15b. CHB data: Number of trials per temperature bracket, for IMO type-approved treatment systems, for temperatures of 0-5°C.

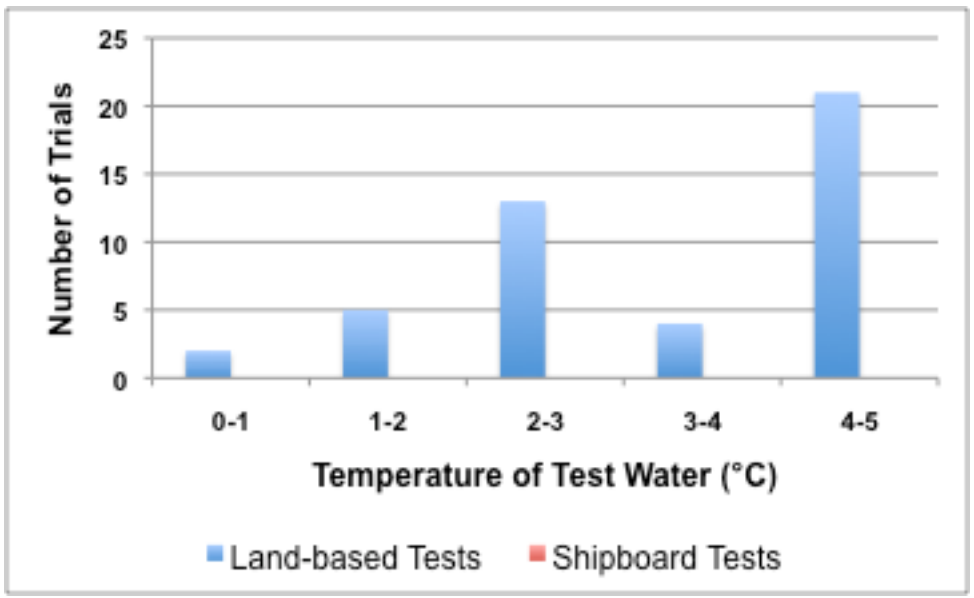
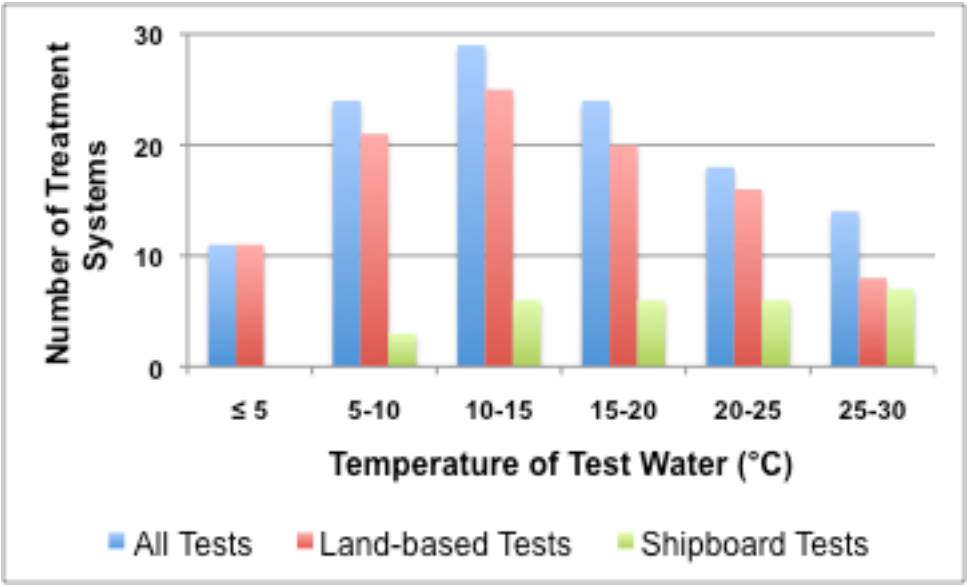


Figure 16. CHB data: Number, per temperature bracket, of IMO type-approved treatment systems with at last one trial in the overall data set.



Figures 17 to 20 show the average and minimum reductions (or in a number of trials, increases) in CHBs over different temperature brackets. As with zooplankton and phytoplankton, no type of treatment appears to be superior to the others, and no trend in performance relative to temperature is demonstrated. However, the data do show a large range of treatment performance. In Figure 17, the average performance of 15 Biocide-based treatment systems ranges over 4-5 orders of magnitude, the performance of 17 UV systems ranges over nearly 6 orders of magnitude, and the performance of 7 UV+ systems ranges over nearly 4 orders of magnitude.

Figure 17. Average and smallest CHB reductions by treatment system and treatment type, for all trials. Treatment systems with ≥5 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

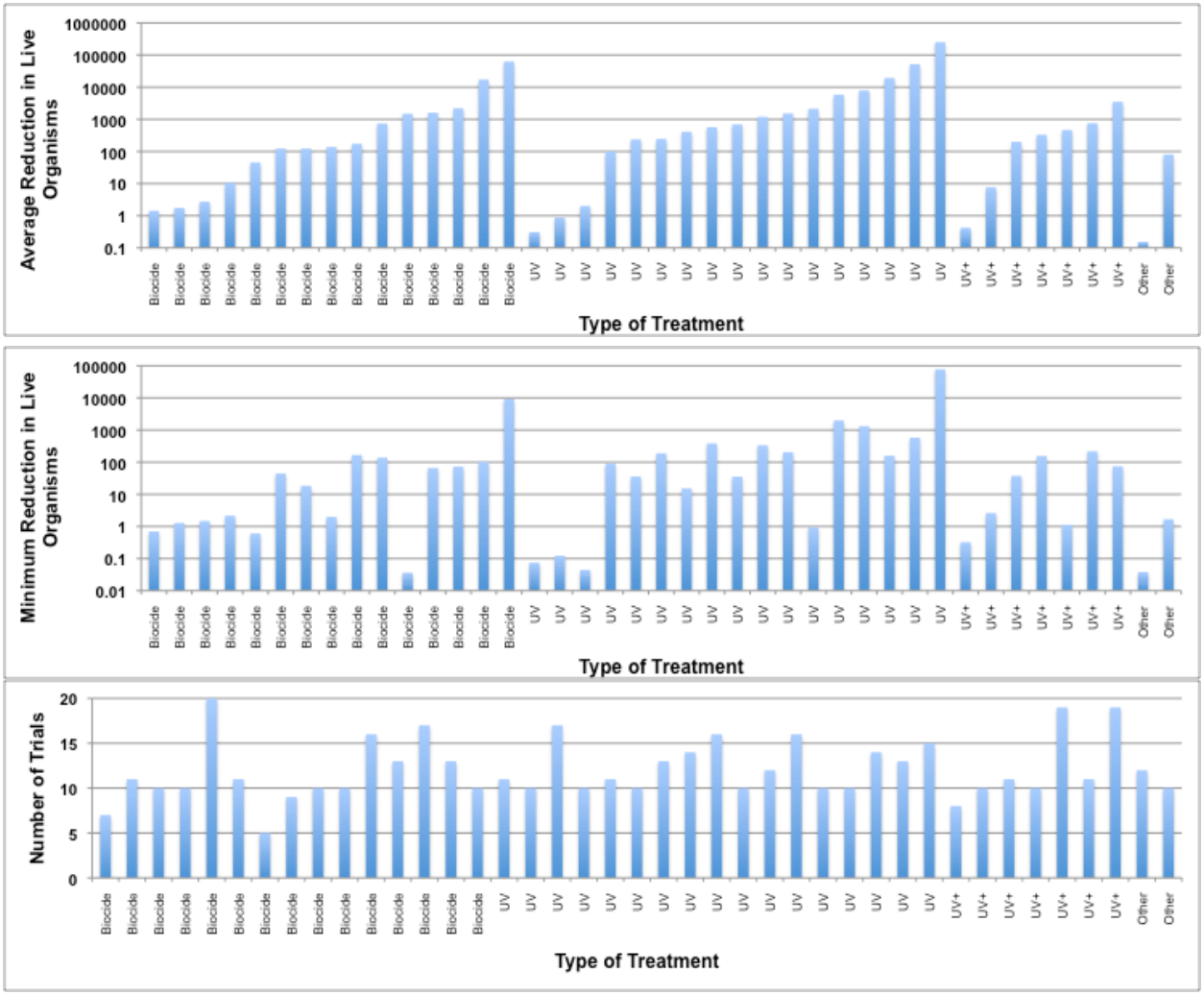


Figure 18. Average and smallest CHB reductions by treatment system and treatment type, for trials with intake water temperatures between 0° and 5°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

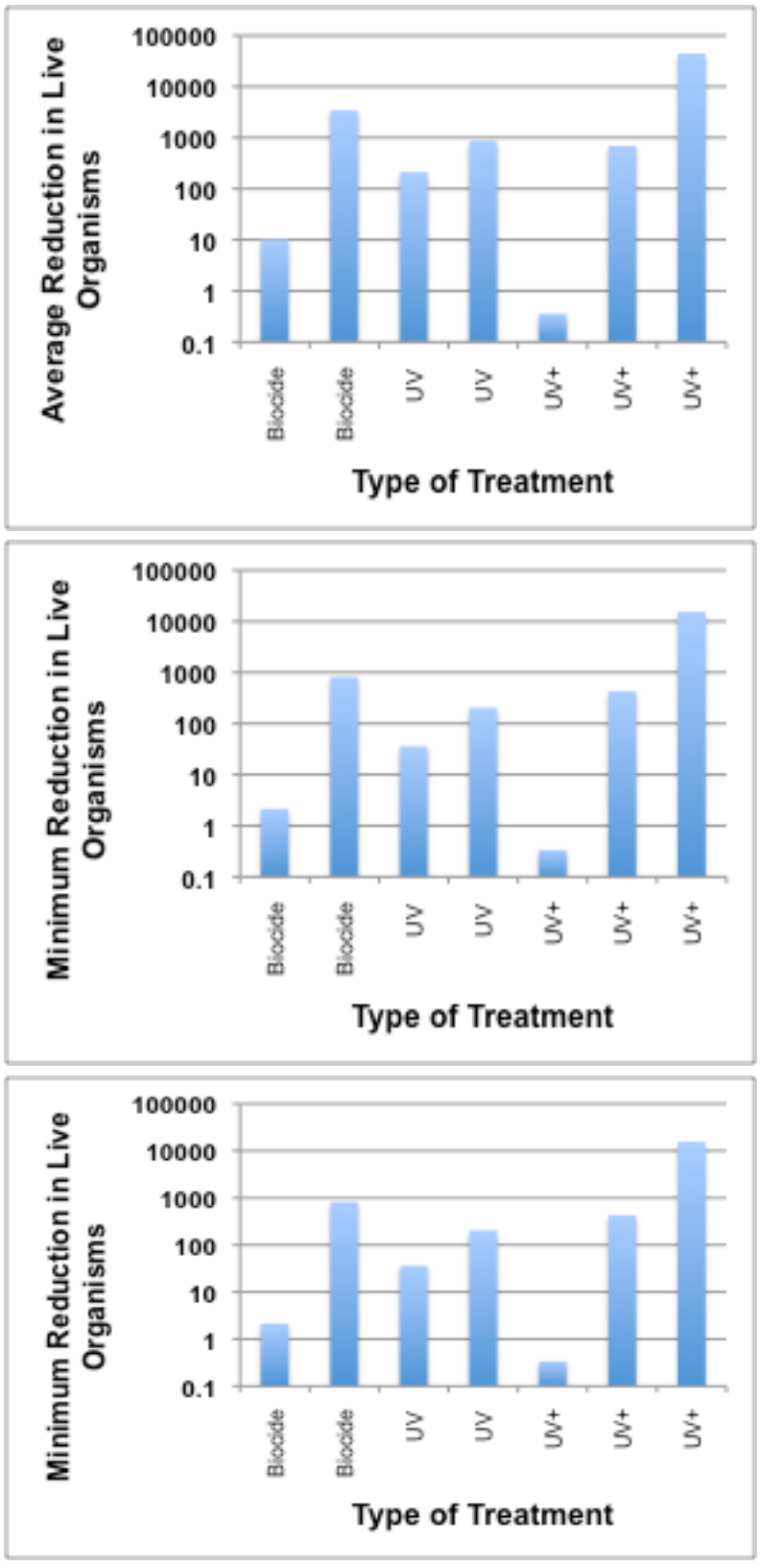


Figure 19. Average and smallest CHB reductions by treatment system and treatment type, for trials with intake water temperatures between 5° and 10°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.

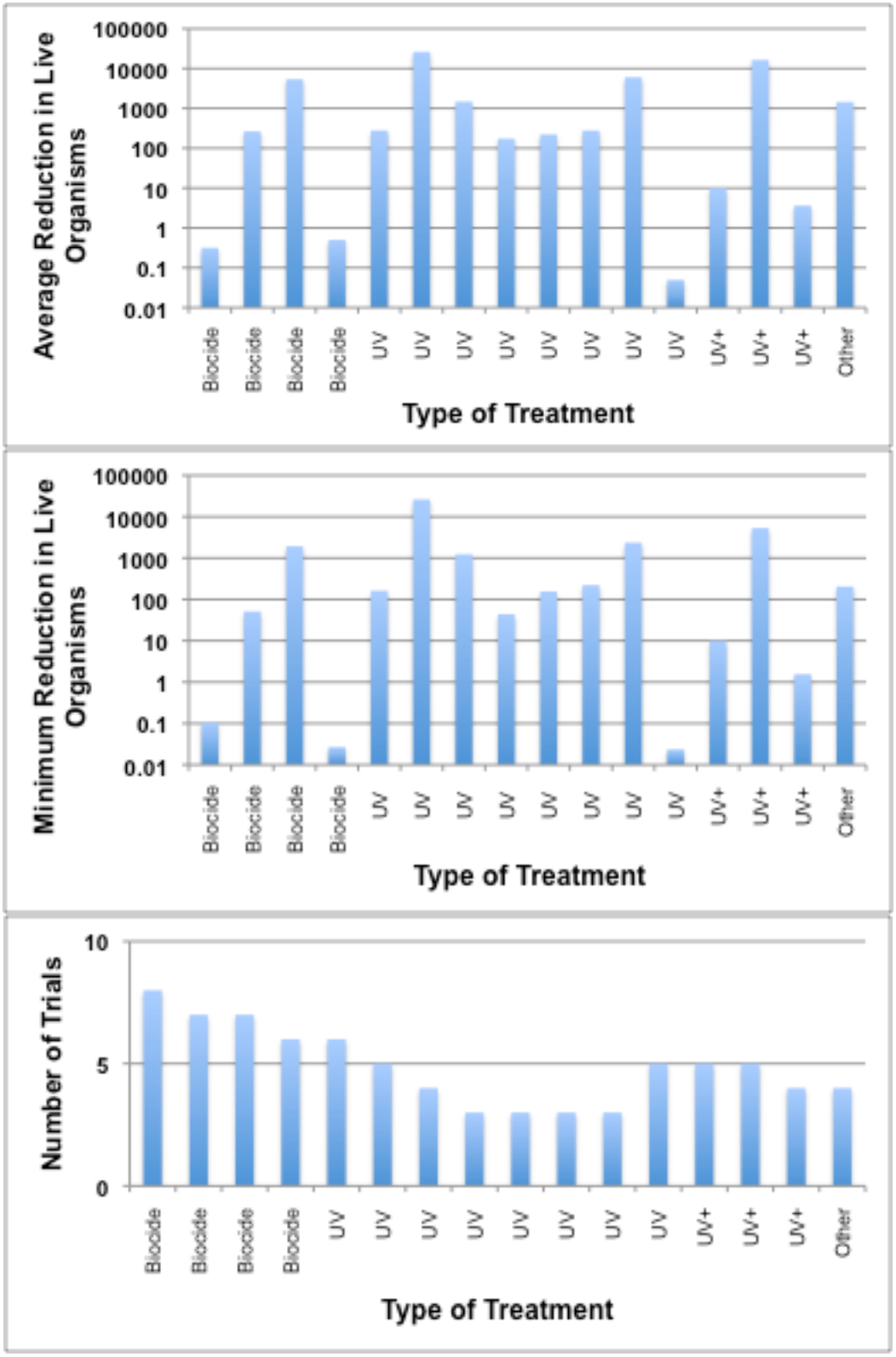
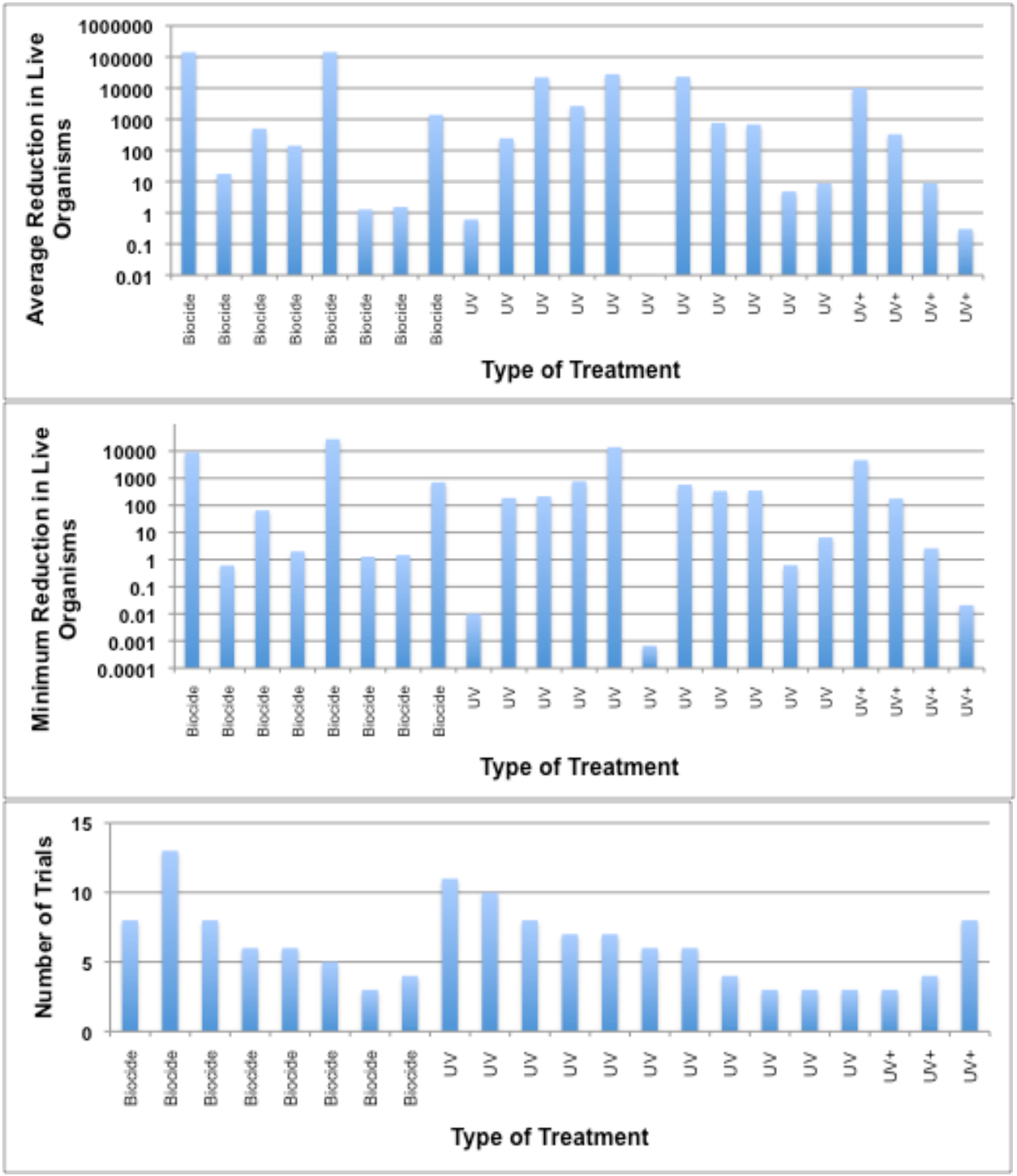


Figure 20. Average and smallest CHB reductions by treatment system and treatment type, for trials with intake water temperatures between 10° and 15°C. Treatment systems with ≥3 trials included. The treatment systems are arranged in the same left-to-right order in the 3 graphs in this figure.



Treatment Processes and Low Temperature

Various studies have found that for some treatment processes used in shipboard ballast water treatment systems, operation is impaired and/or effectiveness declines in colder water (Drillet *et al.* 2013). With reduced efficacy, treatment may require a higher dosage or longer contact time to achieve the same effect. These issues have been raised regarding treatment with chlorine (Weber & Levine 1994; LeChevallier & Au 2004), chloramine (Weber & Levine 1994), chlorine dioxide (Gregg & Hallegraeff 2007), ozone (Hirata *et al.* 2001), peracetic acid (Gregg & Hallegraeff 2007), menadione (Gregg & Hallegraeff 2007), and UV radiation (Abu-Ghararah 1997; Dehgani *et al.* 2013).

In cold water, some types of deoxygenation treatment need a longer period of time to reduce oxygen concentrations to target levels (Mesbahi 2004; De Lafontaine & Despatie 2014), and organism mortality can also be reduced in cold water. Electrochlorination is reported to function optimally in seawater warmer than 15°C, to produce less oxidant at 10-15°C, and often to not function at all in water colder than 5°C (Vom Baur 2013). Treatment processes that rely in part on heating the ballast water will likely be less effective when applied to cold ballast water on a voyage through cold water. In addition, harsh conditions in the Arctic could create other problems for ballast water treatment systems, such as freezing pipes or valves, and rough seas damaging UV lamps and reactors (Jing *et al.* 2012).

Low Temperature Limits on Treatment System Operation

I reviewed type approval certificates from several countries, IMO type approval reports, US Coast Guard AMS acceptance letters, and manufacturers' descriptive and technical materials for information on low temperature operating limits of ballast water treatment systems. Aside from reported limits around -2°C (the freezing point of seawater), I found reported limits for 14 of the 72 treatment systems that have been type-approved under IMO guidelines. The reason for the limit is generally not given.

- Six treatment systems are limited to intake water warmer than 0°C. Four of these use UV radiation, one of which additionally generates ozone; at least 3 of these use low pressure lamps. One system injects chlorine dioxide and one uses in-line electrochlorination.
- Three treatment systems that use in-line electrochlorination have lower temperature limits of 2° and 3°C.
- Four electrochlorination treatment systems, at least 3 of which use side-stream electrochlorination, are limited to intake water with temperatures of at least 4°C, 10°C or 15°C. If they are to be operated in water with lower temperatures, the intake water must be heated or augmented with water from the ship's cooling system or dump condenser system.
- Documents state that one ozone-injecting treatment system has an operating range of 20.35° to 25.49°C. The AMS acceptance letter for the treatment system says that it "is not accepted as an AMS for use in US Arctic waters," a statement that is not found in any other AMS acceptance letter. Examination of the type certificate for this treatment system reveals that it was tested in 20.35° to 25.49°C water; I suspect that the entry of this same range onto the line labeled "Permitted range of temperature of seawater inlet" was probably an error (a ballast water treatment system with only a 5°C temperature range will find little use), and the subsequent statement in the AMS letter was simply a compounding of that error.

Comments and Recommendations

Summary of Results and Analysis

Typical surface water temperatures in the Arctic Core region ranged from nearly -2° to 5°C and Subarctic region temperatures ranged up to 13°C prior to any significant warming effects from anthropogenic climate change. In recent decades surface water temperatures in some parts of the Arctic have increased by as much as 2°C.

Arctic waters could be impacted by non-native species arriving in ballast water from the Antarctic (though the likelihood of significant transport of ballast water from Antarctic waters to the Arctic seems low); by species transported from one part of the Arctic to another, including transport from coastal waters of the mainland to islands; and by transport of species from warmer regions, especially northern hemisphere Boreal/Cold Temperate waters. Ballast treatment systems work in a variety of modes: some operate only during ballast uptake, some operate on both uptake and discharge, and a few operate either during the voyage or only at discharge. Thus, depending on the ballast water source region and the type of treatment system installed on a ship, critical operations to prevent the introduction via ballast water of nonnative species could occur in the Arctic, or outside of the Arctic in warm or cold water. In particular it is not known how likely it is that critical ballast treatment operations will need to be conducted in the coldest Arctic waters between -2° and 0°C.

Studies of treatment processes, and the operational limits included in some type approval certificates, indicate that some of the processes used in shipboard ballast water treatment systems may not operate correctly in cold Arctic waters or may function with reduced effectiveness. Given this, it is somewhat surprising that the analyses of test data reported here did not find any trends toward reduced effectiveness at lower temperatures (see Figures 1, 9 and 14; also comparisons among Figures 6-8, 11-13 and 18-20). Three factors may, in part, explain this: (1) no trials were conducted in the coldest water, below 0.7°C; (2) the enormous range of variation in treatment performance may mask any trends; and (3) the variation in test procedures and conditions, discussed briefly below, may also mask any trends.

Treatment System Testing; and Access to Performance Data

The transport of organisms in ballast water poses risks to ecosystems, to public health, and to economic activities. Actions to reduce those risks impose costs on the shipping industry. Accurate assessment of the capabilities of ballast water treatment systems, and general dissemination of that information, would serve the interests of both the shipping industry and the public.

Unfortunately, our current system falls short. The present analysis was conducted on test data assembled through substantial effort over several years. While some test data were obtained on nearly all the type-approved treatment systems, the extent and quality of that data varies greatly. In some cases fairly complete descriptions of the test procedures, supporting documentation, and extensive data (sometimes including copies of the original data logs) are available; in other cases there are only summaries of some of the results, and little else; and for some type-approved systems, there is no test data available at all.

Because there has been no requirement that test data be released,[13] it has been difficult for ship owners to assess the relative capabilities of the treatment systems in order to make informed purchases, and many lack assurance that the treatment systems will perform as promised across the range of environmental conditions that a ship is likely to encounter. The failure to release these data has also posed an obstacle to agencies, civic organizations and members of the public seeking to assess the effectiveness of ballast water policy, and to researchers trying to identify and resolve problems with the testing and approval process. I would recommend that the IMO, or any institution responsible for ballast water policy at a national or regional level, require the public release of all test data and supporting information used in assessing treatment system performance in order to grant type approval, as a condition of operating those systems to treat ballast water for discharge within the relevant jurisdiction.

Beyond the lack of public access to the data, there are numerous unresolved problems and shortcomings in the testing and type approval process (*e.g.* SAB 2011; Cohen & Dobbs 2015; Bergman 2017). Underlying or magnifying many of these is a lack of consistency in the way the existing test protocols are interpreted by the laboratories conducting the tests. This is in part a product of the decentralized approval process, wherein test results are reviewed and approvals are issued by IMO's individual Member States, rather than by a single body reporting to IMO. Without a single final authority it's difficult (or probably impossible) to establish consistent test procedures.[14]

In this context, consider the challenge facing a member of the public, a researcher or a ship owner trying to understand how effective a ballast water treatment system will be in cold water environments. While the test protocols require that land-based trials be conducted in at least two salinity ranges,[15] there is no comparable requirement regarding water temperature.[16] The

---

[13] The earlier IMO guidelines (IMO 2005, 2008) said nothing about releasing test data to the public. Subsequent resolutions invited but did not require IMO Member States to report to IMO on type approvals, including test data, and instructed the IMO Secretariat to "make such information available by an appropriate means" (IMO 2008b, 2013). The test data and supporting information made available by the Secretariat on individual treatment systems ranges from extensive to none. The recently amended guidelines and proposed code direct Member States granting type approval to obtain detailed test data and supporting information from applicants and to "redact commercially sensitive information (information that is proprietary and not related to the BWMS performance) and make all other information available to interested parties" (IMO 2016 at Part 2, §2.7; IMO 2017 at Part 2, §§2.56-2.60). These provisions contain no deadlines for responding to requests for this information or making it available to interested parties, no review of redaction decisions, no recourse if requests are not responded to or information is not provided, and no enforcement mechanism. Experience has shown that some manufacturers may consider *all* ballast water treatment system test data submitted for government review and approval to be proprietary and confidential, and some governments may honor this; and that aside from that, requests to governments for ballast water treatment system test data may languish for years with no response, despite laws requiring prompt response and release of such data. While the recent guidelines and proposed code are a step forward in that they provide a clear expression of an intent to make such data available, it remains to be seen how well this works in practice without deadlines, review or enforcement.

[14] IMO did set up a central authority to review test data and approve treatment systems with regard to their use of active substances and meeting limits on residual toxicity (known as Basic Approval and Final Approval). It's not clear why IMO does not rely on a central authority to review test data and approve treatment systems with regard to their effectiveness at removing or inactivating organisms in ballast discharges (known as Type Approval).

[15] The 2016 guidelines and proposed code require testing at three salinity ranges for approval to operate at all salinities (IMO 2016 at Part 2, §§2.4.20, 2.4.22, IMO 2017 at Part 2, §§2.29, 2.31).

[16] The 2008 guidelines only state that a Type Approval Certificate should be issued for the specific application for which the treatment systems is approved, which could include restriction to a specific

series of trials needed for type approval could be conducted over a wide range of temperatures, or they could all be conducted at a single temperature; either is equally acceptable under the test protocols and approval requirements. A manufacturer, by selecting the land-based test facility and the vessel and voyages on which shipboard trials will be conducted, can essentially select the temperature the treatment system will be tested at, which is one way these tests can be "gamed"— completely legitimately, under the test rules.[17]

What's the alternative? A simple approach would be to require testing at the extremes: conduct some tests in very cold water, near or even below 0°C, and conduct some in very hot water, in the mid to upper thirties, as occurs in some tropical seas. A more sophisticated approach would be to first determine what water temperature, out of the range of temperatures that a ship might encounter, is the most challenging for the treatment system under consideration, and then test the system at that temperature. If a treatment system posts good test results in environmental conditions that are among the most challenging for it, that provides assurance that it will perform well under real-world working conditions.

Consider the obverse of the data presented in Figure 3. Fifteen treatment systems had data from at least one trial conducted with water intake temperatures of 0°-5°C. That means that there are 57 type-approved treatment systems (79% of the total) for which there are no public data available for even a single trial conducted with water colder than 5°C. If testing under challenging or extreme conditions isn't required, then testing under challenging or extreme conditions won't generally be done.

The 2016 guidelines should bring some limited improvement to the situation. Though not specifically requiring testing at low temperatures, nor even assessment of any kind for temperatures below 0°C, the guidelines would at least appear to require the type-approving party to state the temperature range over which it has determined the approved treatment system can operate effectively, and to provide documentation to support that determination. How well this will work in practice remains to be seen.

Treatment System Performance

The analysis also did not find that any of the general types of treatment consistently outperformed the other types of treatment (Figures 5-8, 10-13 and 17-20). What is clear from these data is that there is an enormous range in performance by shipboard ballast water treatment systems, both across all treatment types and within major treatment types, and that

---

temperature regime (IMO 2008 at §6.2). The 2016 guidelines additionally state that the effective performance of a treatment system should be assessed over a temperature range of 0°-40°C (2°-40°C for fresh water) by land-based, shipboard, laboratory or bench-scale testing or by the use of existing data or models (IMO 2016 at Part 2, §2.5). However, even if testing is selected as the method of assessment, this provision does not specifically require that tests be conducted at any particular temperature, and it's unclear what suite of tests will be conducted in such assessments. Nor does this provision indicate any need to even assess the ability of treatment systems to operate effectively in water temperatures below 0°C, such as may be encountered in Arctic waters year-round and Boreal waters in the winter (Sverdrup *et al.* 1942).

[17] Changes included in the 2016 guidelines and proposed code create an additional challenge, by eliminating the required minimum organism concentration in the control discharge or any requirement to measure organism concentrations in the control discharge in shipboard trials. As a result, under the new guidelines it will no longer be possible to assess the degree to which ballast water treatment reduces organism concentrations in shipboard trials. This will be a significant loss of information: shipboard trials provided 29% of the test data analyzed in this paper.

the weakest performances are quite bad. For example, in the best zooplankton trial, treatment of the ballast water reduced organism concentrations more than 300,000 times as much as they were reduced in the worst zooplankton trial (concentrations were reduced by a factor of 5 in the worst trial, and by a factor of 1.6 million in the best trial). For phytoplankton, the best trial did more than 400 million times better than the worst trial (the best trial reduced concentrations by 2.6 million times, the worst trial increased concentrations by 150 times). For CHBs, the best trial did nearly 3 billion (!) times better than the worst trial; in the worst trials, CHB concentrations were considerably *greater* in the treated discharges than in the untreated discharges.[18]

Comparing the best and worst individual trials is probably not the best way to assess this. A more reasonable approach might be to compare the average performance of the best performing treatment system with the average performance of the worst performing treatment system. Restricting this comparison to treatment systems with data from a minimum of 5 trials, for zooplankton the best performing system, on average, did more than 20,000 times better than the worst performing system (the worst performing system, with data from 19 trials, reduced concentrations on average by a factor of 12; the best system, with data from 14 trials reduced concentrations on average by a factor of over 260,000). For phytoplankton, the best system outperformed the worst by over 1 million times. For CHBs, the best system outperformed the worst by more than 1.6 million times.

Remember that the treatment systems in the analysis, from the worst-performing to the best, have all been awarded type approval. Under IMO's current rules, any of these treatment system can be used to meet IMO's requirements. The significance of the enormous differences in performance among these systems is that probably the most effective single thing that could be done to protect Arctic waters from ballast water invasions would be to require ships to install and operate one of the better performing treatment systems, instead of allowing them to install the worst ones.

Acknowledgments

This paper contains discussion regarding the importance of, and existing limitations on, the public release of ballast water treatment system test data. I want to acknowledge here those individuals, companies and institutions that were especially helpful in providing or releasing data that I used in the analysis. However, I also note where data have not been released, to illustrate difficulties that interested parties face in trying to obtain information on the performance of ballast water treatment systems.

Manufacturers of Ballast Water Treatment Systems: I'd like to thank Mike Lee, Mi Eun Oh and Dong Hyun Shon at Techcross, and Lawrence Lee and Jiandong Shen at Elite Marine, for providing reports and answering my questions. However, most of my requests to manufacturers for test data and other information either did not receive a response or were rejected on the basis that the information is considered to be confidential.

Test Laboratories: As a rule, the laboratories conducting type approval tests have been unwilling to release test data on a treatment system without explicit permission from the manufacturer, out of concern that the manufacturer may consider such data to be confidential. However, with a few exceptions, most have been willing to provide information and answer

[18] See Cohen & Dobbs (2015) regarding shipboard treatment turning ballast tanks into bacteria incubators.

questions on their test methods. In particular, I'd like to thank those who provided information on the detection limits in their tests—which was essential data needed to assess many of the trials—including Stephanie Delacroix, Yasuwo Fukuyo, Stephan Gollasch, Pung-Guk Jang, Young-soo Kim, Louis Peperzak, Shunji Sukizaki, Marcel Veldhuis, Qiong Wang and David Wright. In several cases it took these individuals a great deal of work to dig up the necessary information as well as substantial back-and-forth correspondence to clarify the details, and I am grateful for their grace and patience in dealing with me.

Government Agencies and Agents: I obtained a number of test reports through a Public Records Act request to the California State Lands Commission, which had assembled the reports to support regulatory policy decisions. Of 30 requested reports, the Commission provided 27 in their entirety and one with some paragraphs redacted; two were not provided because the manufacturers considered their contents to be confidential trade secrets. I am grateful to staff attorney Joe Fabel for his prompt response to this request, though I continue to believe that government agencies should make public all information they rely on in making policy decisions unless there is an overriding public concern such as national security.

I also obtained a number of test reports through a series of Freedom of Information Act (FOIA) requests to the U.S. Coast Guard. These included some reports that the Coast Guard received in applications for AMS status or had commissioned, a few of which had some material redacted before they were provided to me. However, many other reports and related documents that I requested were never provided, including requests that I initially made more than four years ago. No official response or explanation was ever provided, even though the law requires the agency to provide such a response within 20 working days. Since September of this year there has been a renewed effort by some Coast Guard staff to resolve this, though it is not yet clear whether it will result in my receiving any more of the requested test reports. In September I also filed a FOIA request for the test reports that the Coast received in applications for US type approval. As of this date I have not received any reports nor any response.

The Bundesamt für Seeschifffahrt und Hydrographie (BSH) manages the IMO type approval of ballast water treatment systems for Germany. It has made available on its website the test reports for the treatment systems it has granted type approval to. This is a useful service that other type-approving agents could emulate.

Literature Cited

Abu-Ghararah ZH. 1997. Effect of temperature on the kinetics of wastewater disinfection using ultraviolet radiation. Journal of King Saud University - Engineering Sciences 9: 171-186.

AMAP. 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Council, Arctic Monitoring and Assessment Programme, Oslo, Norway.

ARMS. 2017. The Arctic Register of Marine Species (ARMS) website, at http://www.marinespecies.org/arms, accessed in October 2017.

Bomford M, Barry SC, Lawrence E. 2010. Predicting establishment success for introduced freshwater fishes: a role for climate matching. Biological Invasions 12(8): 2559-2571.

Carlton JT, Reid DM, van Leeuwen H. 1995. Shipping Study. The Role of Shipping in the Introduction of Nonindigenous Aquatic Organisms to the Coastal Waters of the United States (other than the Great Lakes) and an Analysis of Control Options. Report No. CG-D-11-95, US Coast Guard, Groton, CT and US Department of Transportation, Washington, DC, USA.

Carlton JT. 1985. Transoceanic and interoceanic dispersal of coastal marine organisms: the biology of ballast water. Oceanography and Marine Biology 23: 313-371.

Chan FT, Briski E, Bailey SA, MacIsaac HJ. 2014. Richness–abundance relationships for zooplankton in ballast water: temperate versus Arctic comparisons. ICES Journal of Marine Science 71(7): 1876-1884.

Chan FT, Bronnenhuber JE, Bradie JN, Howland KL, Simard N, Bailey SA. 2012. Risk Assessment for Ship-mediated Introductions of Aquatic Nonindigenous Species to the Canadian Arctic. Canadian Science Advisory Secretariat Research Document 2011/105, Fisheries and Oceans Canada, Burlington, ON, Canada.

Christiansen JS, Reist JD. 2013. Chapter 6. Fishes. Pages 192-245 in: Meltofte H (ed.), Arctic Biodiversity Assessment; Status and Trends in Arctic Biodiversity. Arctic Council, Conservation of Arctic Flora and Fauna, Akureyri, Iceland.

Christie P, Sommerkorn M. 2012. RACER: Rapid Assessment of Circum-arctic Ecosystem Resilience, 2nd Edition. World Wildlife Fund, Global Arctic Programme, Ottawa, ON, Canada.

Cohen AN, Carlton JT. 1998. Accelerating invasion rate in a highly invaded estuary. Science 279: 555-558.

Cohen AN, Dobbs FC, Chapman PM. 2017. Revisiting the basis for US ballast water regulations. Marine Pollution Bulletin 118: 348-353.

Cohen AN, Dobbs FC. 2015. Failure of the public health testing program for ballast water treatment systems. Marine Pollution Bulletin 91: 29-34.

Cohen AN. 2016. White Paper on Ship-mediated Bioinvasions in the Arctic: Pathways and Control Strategies. A report for Friends of the Earth U.S., Berkeley, CA, USA. Published as: Ship-mediated Bioinvasions in the Arctic: Pathways and Control Strategies, Submitted by FOEI, MEPC 69/INF.17 (9 February 2016), International Maritime Organization, Marine Environment Protection Committee, London, UK.

Darrigran G, Pastorino G. 1995. The recent introduction of a freshwater Asiatic bivalve, *Limnoperna fortunei* (Mytilidae) into South America. Veliger 38(2): 171-175.

De Rivera CE, Steves BP, Fofonoff PW, Hines AH, Ruiz GM. 2011. Potential for high- latitude marine invasions along western North America. Diversity and Distributions. 17(6): 1198-1209.

Dehghani MH, Jahed GR, Zarei A. 2013. Investigation of low-pressure ultraviolet radiation on inactivation of Rhabitidae nematode from water. Iranian Journal of Public Health 42(3): 314-319.

Drillet G, Schmoker C, Trottet A, Mahjoub MS, Duchemin M, Andersen M. 2013. Effects of temperature on type approval testing of ballast water treatment systems. Integrated Environmental Assessment and Management 9(2): 192-195.

Ekman S. 1953. Zoogeography of the Sea. Sidgwick & Jackson, Ltd., London, UK.

Epstein PR, Ford TE, Colwell RR. 1993. Marine ecosystems. Lancet 342: 1216-1219.

Fabry VJ, McClintock JB, Mathis JT, Grebmeier JM. 2009. Ocean acidification at high latitudes: the bellwether. Oceanography 22(4): 160-171.

Gjershaug JO, Rusch GM, Oberg S, Qvenild M. 2009. Alien Species and Climate Change in Norway: an Assessment of the Risk of Spread Due to Global Warming. NINA Report 468, Norwegian Institute for Nature Research, Trondheim, Norway.

Glasby TM, Connell SD, Holloway MG, Hewitt CL. 2007. Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? Marine Biology 151: 887-895.

Gregg MD, Hallegraeff GM. 2007. Efficacy of three commercially available ballast water biocides against vegetative microalgae, dinoflagellate cysts and bacteria. Harmful Algae 6(4): 567-584.

Hallegraeff GM, Bolch CJ. 1992. Transport of diatom and dinoflagellate resting spores in ships' ballast water: implications for plankton biogeography and aquaculture. Journal of Plankton Research 14(8): 1067-1084.

Hallegraeff GM. 1998. Transport of toxic dinoflagellates via ships' ballast water: bioeconomic risk assessment and efficacy of possible ballast water management strategies. Marine Ecology Progress Series 168: 297-309.

Harbison GR, Volvik SP. 1994. The ctenophore, *Mnemiopsis leidyi*, in the Black Sea: a holoplanktonic organism transported in the ballast water of ships. Pages 25-36 in: Nonindigenous Estuarine and Marine Organisms (NEMO), Proceedings of the Conference, US Department of Commerce, NOAA, Seattle, WA, USA.

Hegseth EN, Sundfjord A. 2008. Intrusion and blooming of Atlantic phytoplankton species in the high Arctic. Journal of Marine Systems 74: 108-119.

Hirata T, Shimura A, Morita S, Suzuki M, Motoyama N, Hoshikawa H, Moniwa T, Kaneko M. 2001. The effect of temperature on the efficacy of ozonation for inactivating *Cryptosporidium parvum* oocysts. Water Science and Technology 43: 163-166.

IMO. 1991. International Guidelines for Preventing the Introduction of Unwanted Aquatic Organisms and Pathogens from Ships' Ballast Water and Sediment Discharges. Resolution MEPC 50(31), adopted 4 July 1991, International Maritime Organization, Marine Environment Protection Committee, London, UK.

IMO. 2004. International Convention for the Control and Management of Ships' Ballast Water and Sediments, 2004. International Conference on Ballast Water Management for Ships, International Maritime Organization, London, UK.

IMO. 2005. Guidelines for Approval of Ballast Water Management Systems (G8). Resolution MEPC.125(53), International Maritime Organization, Marine Environmental Protection Committee, London, UK.

IMO. 2008a. Guidelines for Approval of Ballast Water Management Systems (G8). Resolution MEPC.174(58), International Maritime Organization, Marine Environmental Protection Committee, London, UK.

IMO. 2008b. Information Reporting on Type Approved Ballast Water Management Systems. Resolution MEPC.175(58), adopted 10 October 2008. International Maritime Organization, Marine Environmental Protection Committee, London, UK.

IMO. 2013. Information Reporting on Type Approved Ballast Water Management Systems. Resolution MEPC.228(65), adopted 17 May 2013. International Maritime Organization, Marine Environmental Protection Committee, London, UK.

IMO. 2016. 2016 Guidelines for Approval of Ballast Water Management Systems (G8). Resolution MEPC.279(70), International Maritime Organization, Marine Environmental Protection Committee, London, UK.

IMO. 2017. Code for Approval of Ballast Water Management Systems (BWMS Code). Draft MEPC Resolution, MEPC 71/17/Add.1 Annex 5, International Maritime Organization, Marine Environmental Protection Committee, London, UK.

Jing L, Chen B, Zhang B, Peng H. 2012. A review of ballast water management practices and challenges in harsh and arctic environments. Environmental Reviews 20(2): 183-108.

Josefson AB, Mokievsky V. 2013. Chapter 8. Marine invertebrates. Pages 276-309 in: Meltofte H (ed.), Arctic Biodiversity Assessment; Status and Trends in Arctic Biodiversity. Arctic Council, Conservation of Arctic Flora and Fauna, Akureyri, Iceland.

Kerckhof F, Degraer S, Norro A, Rumes B. 2011. Offshore intertidal hard substrata: a new habitat promoting non-indigenous species in the Southern North Sea: an exploratory study. Pages 27-37 in: Degraer S, Brabant R, Rumes B (eds.) Offshore Wind Farms in the Belgian Part of the North Sea: Selected Findings from the Baseline and Targeted Monitoring. Royal Belgian Institute of Natural Sciences, Brussels, Belgium.

Khon VC, Mokhov II, Latif M, Semenov VA, Park W. 2010. Perspectives of Northern Sea Route and Northwest Passage in the twenty-first century. Climatic Change 100(3-4): 757-768.

LeChevallier MW, Au KK. 2004. Water Treatment and Pathogen Control: Process Efficiency in Achieving Safe Drinking Water. Published on behalf of the World Health Organization by IWA Publishing, London, UK.

Locke A, Reid DM, Sprules WG, Carlton JT, van Leeuwen HC. 1991. Effectiveness of mid-ocean exchange in controlling freshwater and coastal zooplankton in ballast water. Canadian Technical Report on Fisheries and Aquatic Sciences, No. 1822, Great Lakes Laboratory for Fisheries and Aquatic Sciences, Canada Centre for Inland Waters, Burlington, ON, Canada.

McCarthy SA, Khambaty FM. 1994. International dissemination of epidemic *Vibrio cholerae* by cargo ship ballast and other nonpotable waters. Applied and Environmental Microbiology 60: 2597-2601.

Mesbahi E. 2004. Latest results from testing seven different technologies under the EU MARTOB project—Where do we stand now? Pages 210-230 in: Matheickal JT, Raaymakers S (eds.), Second International Symposium on Ballast Water Treatment. International Maritime Organisation, London, UK.

Michel C. 2013. Chapter 14. Marine ecosystems. Pages 486-527 in: Meltofte H (ed.), Arctic Biodiversity Assessment; Status and Trends in Arctic Biodiversity. Arctic Council, Conservation of Arctic Flora and Fauna, Akureyri, Iceland.

Miller AW. 2014. Melting sea ice, accelerated shipping, and arctic invasions. Pages 35-55 in: Marine Invasive Species in the Arctic. Nordic Council of Ministers, Copenhagen, Denmark.

Nalepa TH, Schloesser DW (eds.). 2014. Quagga and Zebra Mussels: Biology, Impacts, and Control. 2nd Edition. CRC Press, Taylor & Francis Group. Boca Raton, London and New York.

Nichols FH, Thompson JK, Schemel LE. 1990. Remarkable invasion of San Francisco Bay (California, USA) by the Asian clam *Potamocorbula amurensis*. II. Displacement of a former community. Marine Ecology Progress Series 66: 95-101.

OSPAR. 2000. Quality Status Report 2000: Region I. Arctic Waters. OSPAR Commission for the Protection of the Marine Environment of the North-East Atlantic, London, UK.

Ostenfeld CH. 1908. On the immigration of *Biddulphia sinensis* Grev. and its occurrence in the North Sea during 1903-1907. Meddelelser fra Kommissionen for Havundersogelser, Plankton 1(6):1-25.

PAME. 2013. Large Marine Ecosystems (LMEs) of the Arctic Area: Revision of the Arctic LME Map, 2nd Edition. Arctic Council, Protection of the Arctic Marine Environment, Akureyri, Iceland.

Pearson RG, Dawson TP. 2003. Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? Global Ecology and Biogeography 12(5): 361-371.

Peterson AT. 2003. Predicting the geography of species' invasions via ecological niche modeling. Quarterly Review of Biology 78(4): 419-433.

Renaud PE, Sejr MK, Bluhm BA, Sirenko B, Ellingsen IH. 2015. The future of Arctic benthos: expansion, invasion, and biodiversity. Progress in Oceanography 139: 244-257.

Ricciardi A. 2006. Are modern biological invasions an unprecedented form of global change? Conservation Biology 21(2): 329-336.

SAB. 2011. Efficacy of Ballast Water Treatment Systems: a Report by the EPA Science Advisory Board. U.S. Environmental Protection Agency, Science Advisory Board, Ecological Processes and Effects Committee augmented for the Ballast Water Advisory, Washington, DC, USA.

Sherman K, Hempel G (eds.). 2009. The UNEP Large Marine Ecosystem Report: A Perspective on Changing Conditions in LMEs of the World's Regional Seas. UNEP Regional Seas Report and Studies No. 182, United Nations Environment Programme. Nairobi, Kenya.

Smith LC, Stephenson SR. 2013. New Trans-Arctic shipping routes navigable by midcentury. Proceedings of the National Academy of Sciences. 110(13): E1191-E1195.

Spalding MD, Fox HE, Allen GR, Davidson N, Ferdaña ZA, Finlayson MA, Halpern BS, Jorge MA, Lombana AL, Lourie SA, Martin KD. 2007. Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. BioScience 57(7): 573-583.

Steinacher M, Joos F, Frolicher TL, Plattner GK, Doney SC. 2009. Imminent ocean acidification in the Arctic projected with the NCAR global coupled carbon cycle-climate model. Biogeosciences 6: 515-533.

Sverdrup HU, Johnson MW, Flemming RH. 1942.The Oceans: Their Physics, Chemistry and General Biology. Prentice-Hall, Inc., Englewood Cliffs, New Jersey, USA.

Timmermans ML, Ladd C, Wood K. 2017. Sea surface temperatures. Pages 34-38 in: Richter-Menge J, Overland JE, Mathis JT, Osborne E (eds.), Arctic Report Card 2017, NOAA, Washinton, DC, USA.

USCG. 2012. Standards for living organisms in ships' ballast water discharged in U.S. waters. Final rule; U.S. Coast Guard. U.S. Federal Register 77: 17254-17320.

USEPA. 2010. Generic Protocol for the Verification of Ballast Water Treatment Technology (Version 5.1, Sept. 2010). EPA/600/R-10/146, U.S. Environmental Protection Agency, Washington, DC, USA.

Verna D. 2014. Influences of policy and vessel behavior on the risk of ballast-borne marine species invasions in coastal Alaska. MS Thesis, Alaska Pacific University.

Verna DE, Harris BP, Holzer KK, Minton MS. 2016. Ballast-borne marine invasive species: exploring the risk to coastal Alaska, USA. Management of Biological Invasions 7(2): 199-211.

Vom Baur M. 2013. Ballast water treatment systems – quo vadis? HANSA International Maritime Journal 150(3): 42-49.

Ware C, Berge J, Jelmert A, Olsen SM, Pellissier L, Wisz M, Kriticos D, Semenov G, Kwaśniewski S, Alsos IG. 2016. Biological introduction risks from shipping in a warming Arctic. Journal of Applied Ecology 53: 340-349.

Ware C, Berge J, Sundet JH, Kirkpatrick JB, Coutts ADM, Jelmert A, Olsen SM, Floerl O, Wisz M, Alsos IG. 2014. Climate change, non-indigenous species and shipping: assessing the risk of species introduction to a high-Arctic archipelago. Diversity and Distributions 20: 10-19.

Weber GR, Levine M. 1944. Factors affecting germicidal efficiency of chlorine and chloramine. American Journal of Public Health 34: 719-728.

Appendix A. Supporting Analyses

1. Plots of Treatment system performance vs. water temperature for the Zooplankton Group

The Zooplankton Group consists of organisms that are greater than 50 µm in minimum dimension. System performance is calculated as the organism concentration in the control discharge divided by the organism concentration in the treated discharge. No significant trend in performance relative to temperature is apparent in these plots and regressions.

(a) Physical Separation Processes

Figure A-1. Zooplankton Reduction versus Temperature: Treatment systems with a particle separation stage using Filter and/or Hydrocyclone. y=1803x+38163, n=810, r2=0.0065, p<0.022.

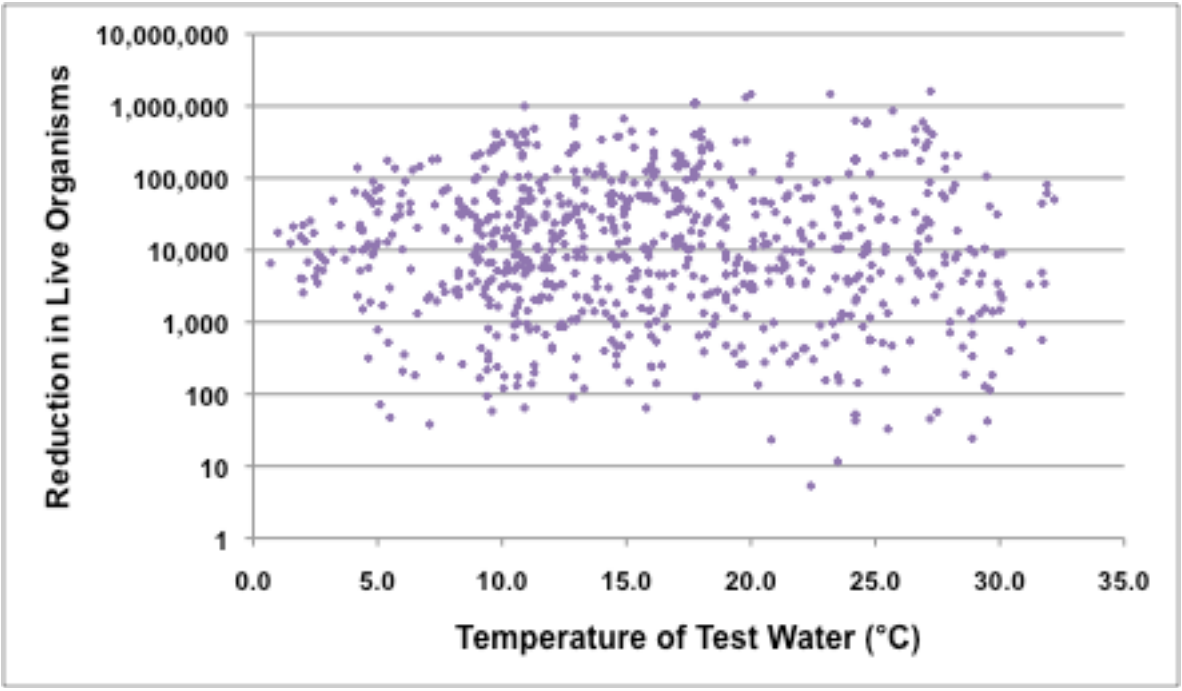


Figure A-2. Zooplankton Reduction versus Temperature: Treatment systems with a particle separation stage using Filter Only (no Hydrocyclone). y=1504x+39622, n=784, r2=0.0054, p<0.04.

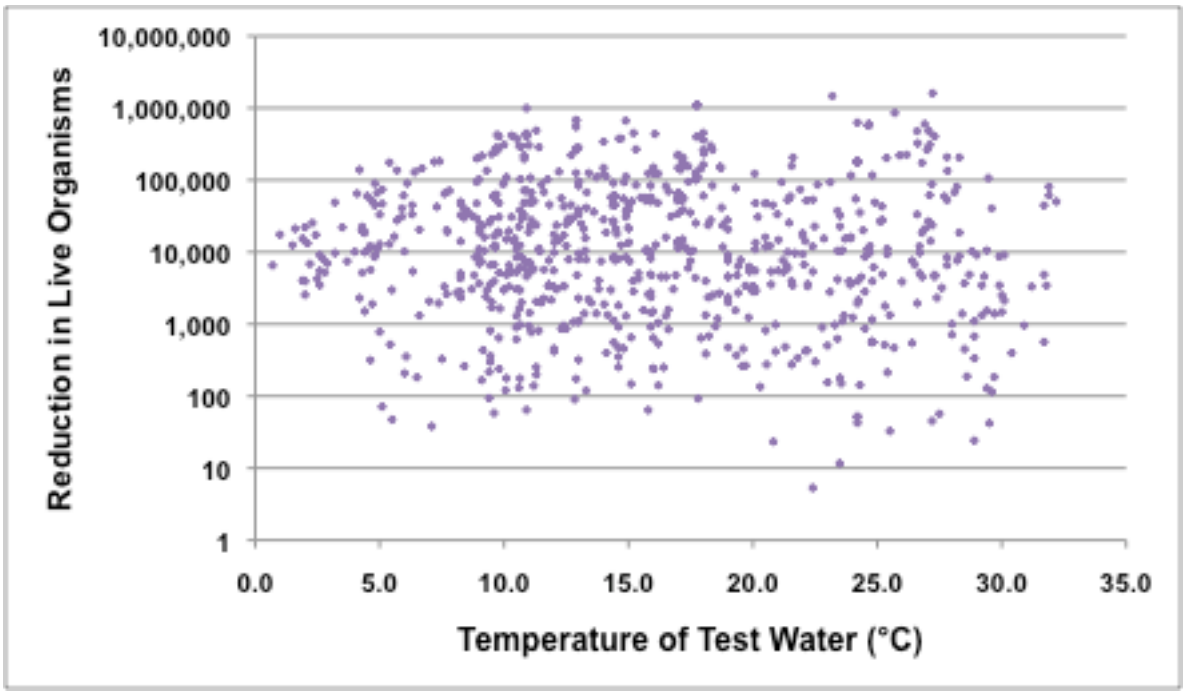


Figure A-3. Zooplankton Reduction versus Temperature: Treatment systems with a particle separation stage using Hydrocyclone, with or without Filter. y=15660x–93990, n=26, r2=0.052, p<0.263.

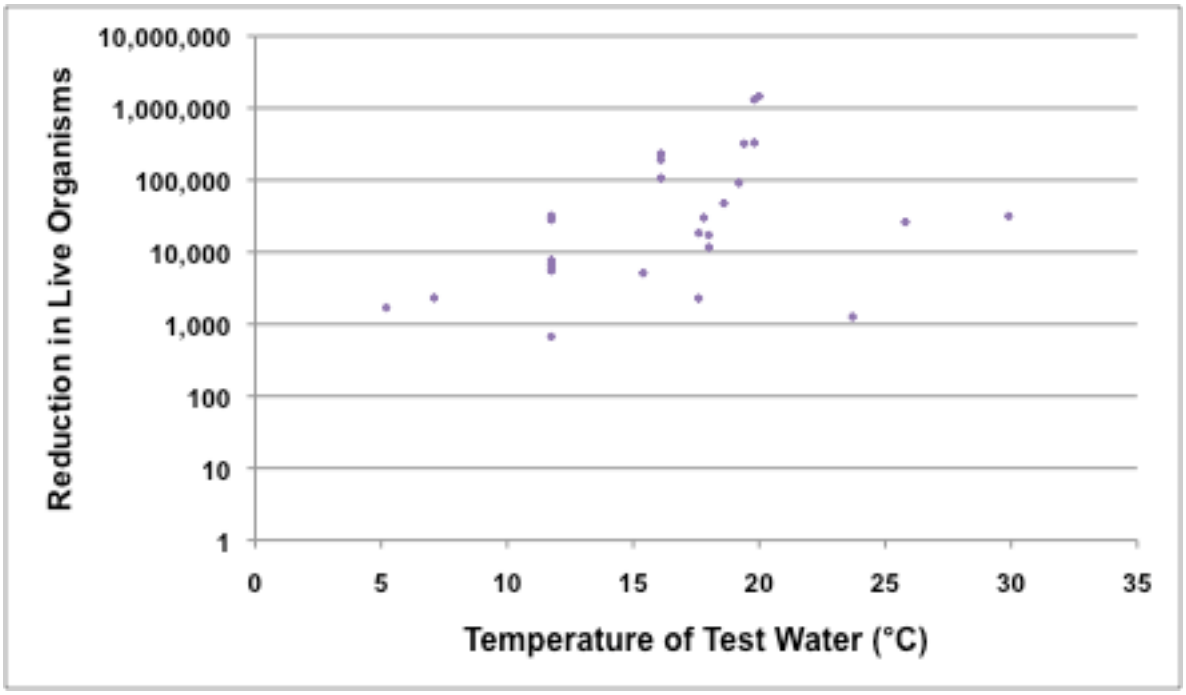


Figure A-4. Zooplankton Reduction versus Temperature: Treatment systems with no particle separation stage (no Filter or Hydrocyclone). y=-3417x+96752, n=175, r2=0.0664, p<0.001.

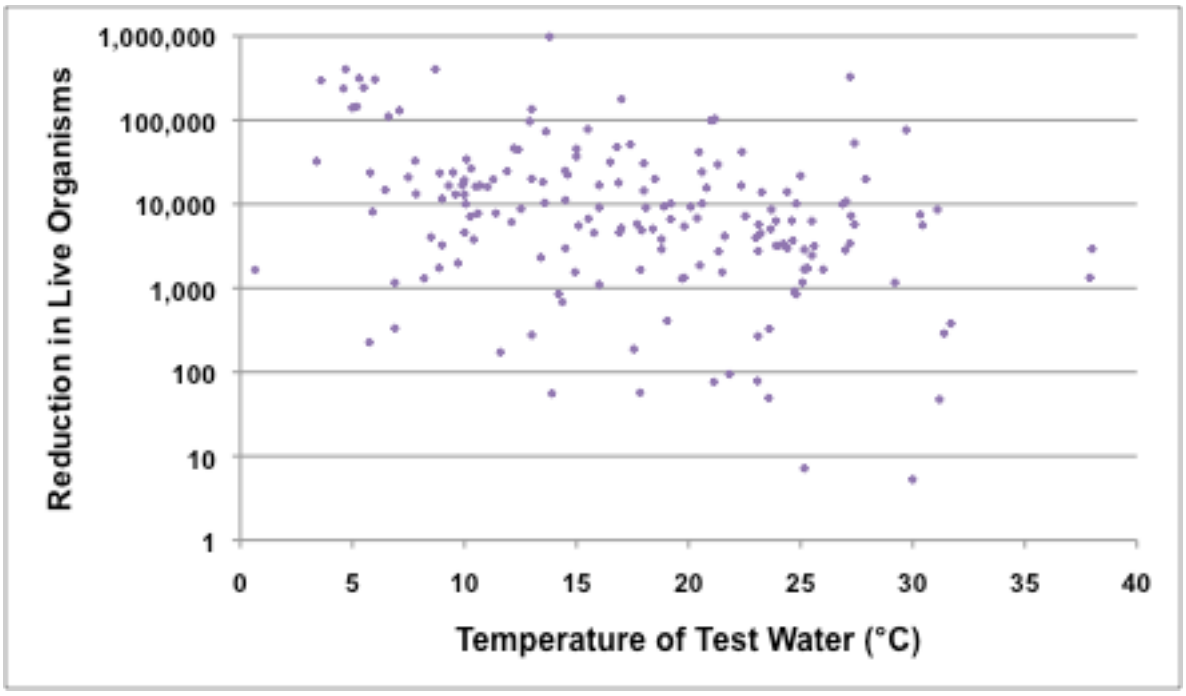
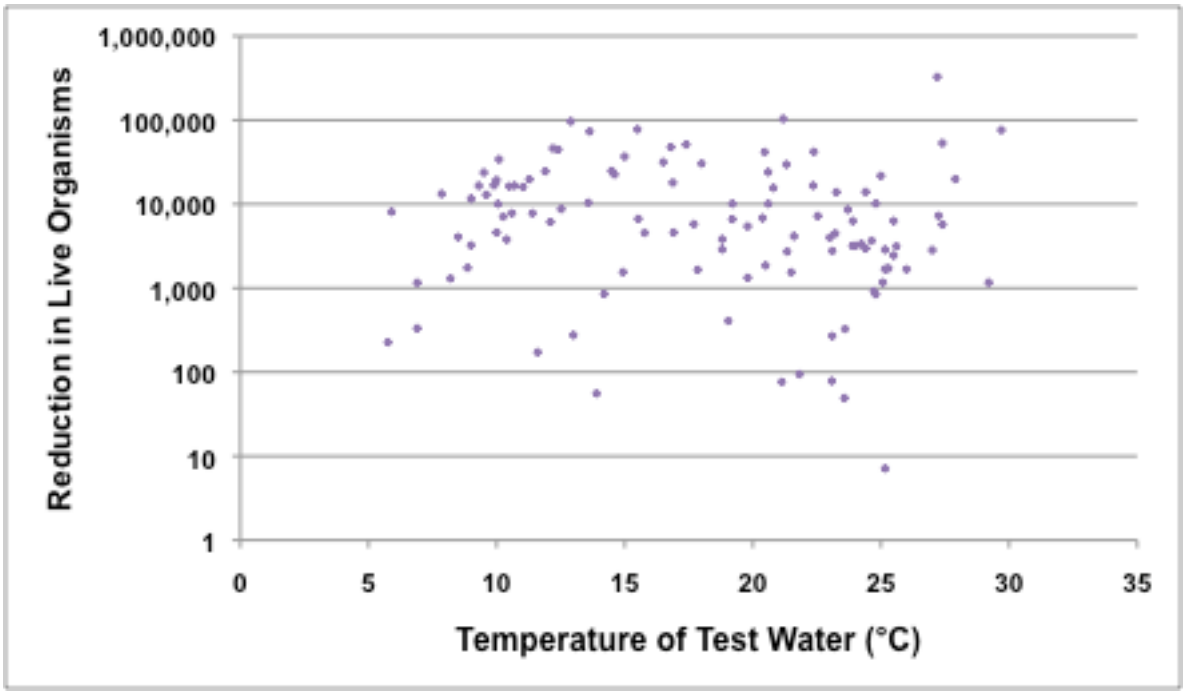


Figure A-5. Zooplankton Reduction versus Temperature: Treatment systems with no particle separation stage, and a disinfection stage that includes use of a biocide. y=411x+9495, n=113, r2=0.0055, p<0.436.



(b) Disinfection Processes

Figure A-6. Zooplankton Reduction versus Temperature: Treatment systems with a disinfection stage that includes use of a biocide. y=2085x+16420, n=358, r2=0.0093, p<0.068.

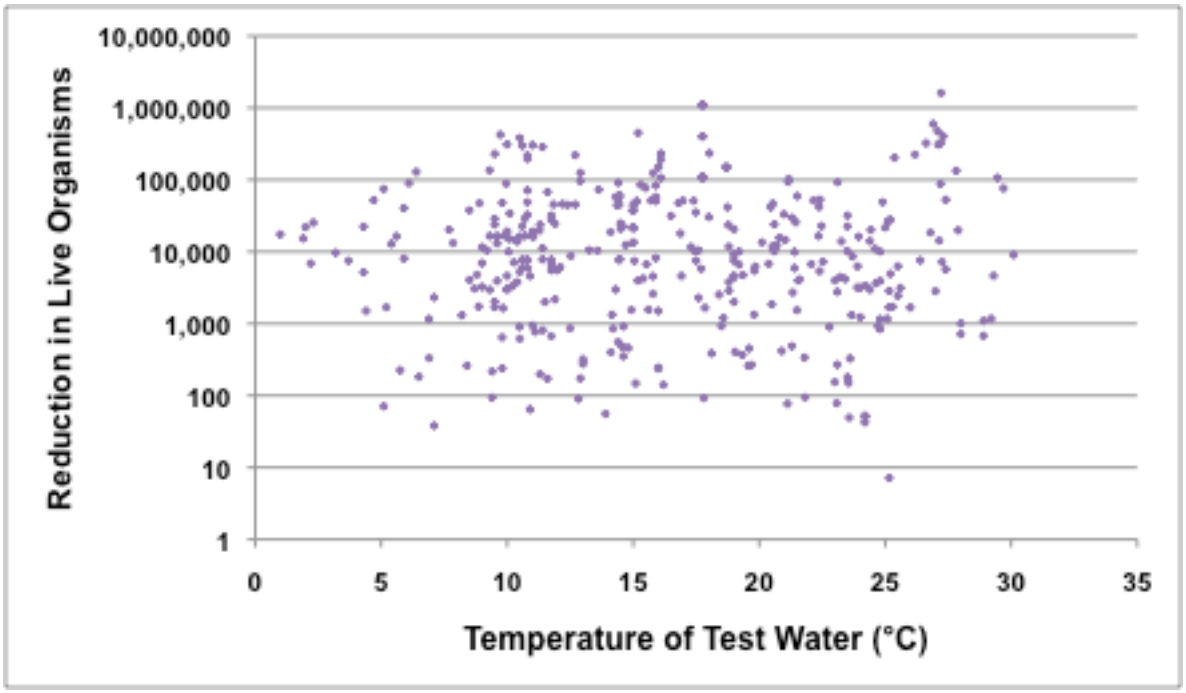


Figure A-7. Zooplankton Reduction versus Temperature: Treatment systems with a disinfection stage that includes use of UV radiation. y=746x+59316, n=553, r2=0.0012, p<0.424.

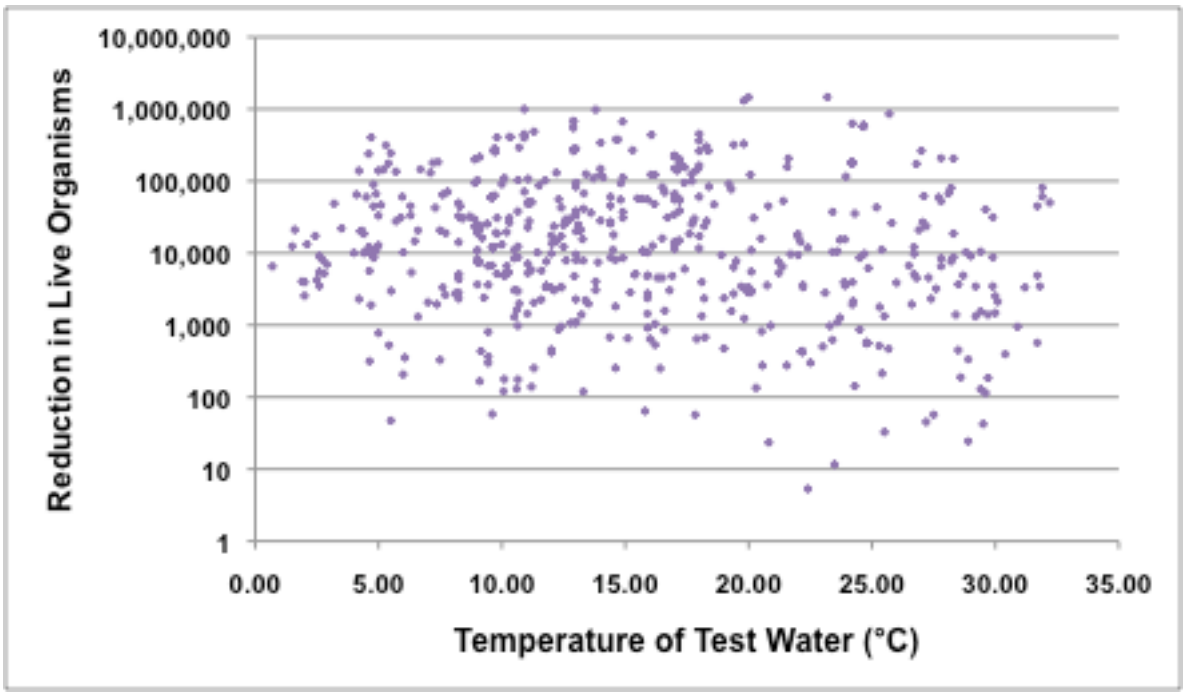
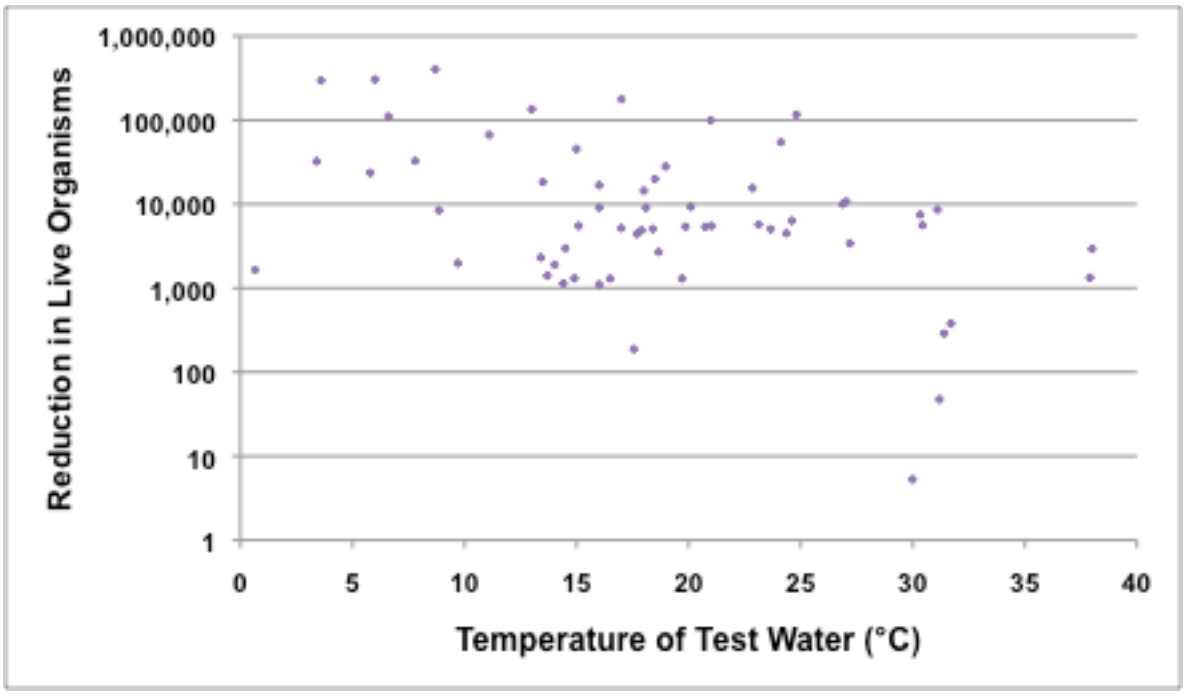


Figure A-8. Zooplankton Reduction versus Temperature: Treatment systems with a disinfection stage that does not use a biocide or UV radiation. y=-3589x+102828, n=61, r2=0.1463, p<0.002.



(c) Salinity

Figure A-9. Zooplankton Reduction versus Temperature: Freshwater Trials (<1 psu). y=-244x+68548, n=50, r2=0.0002, p<0.929.

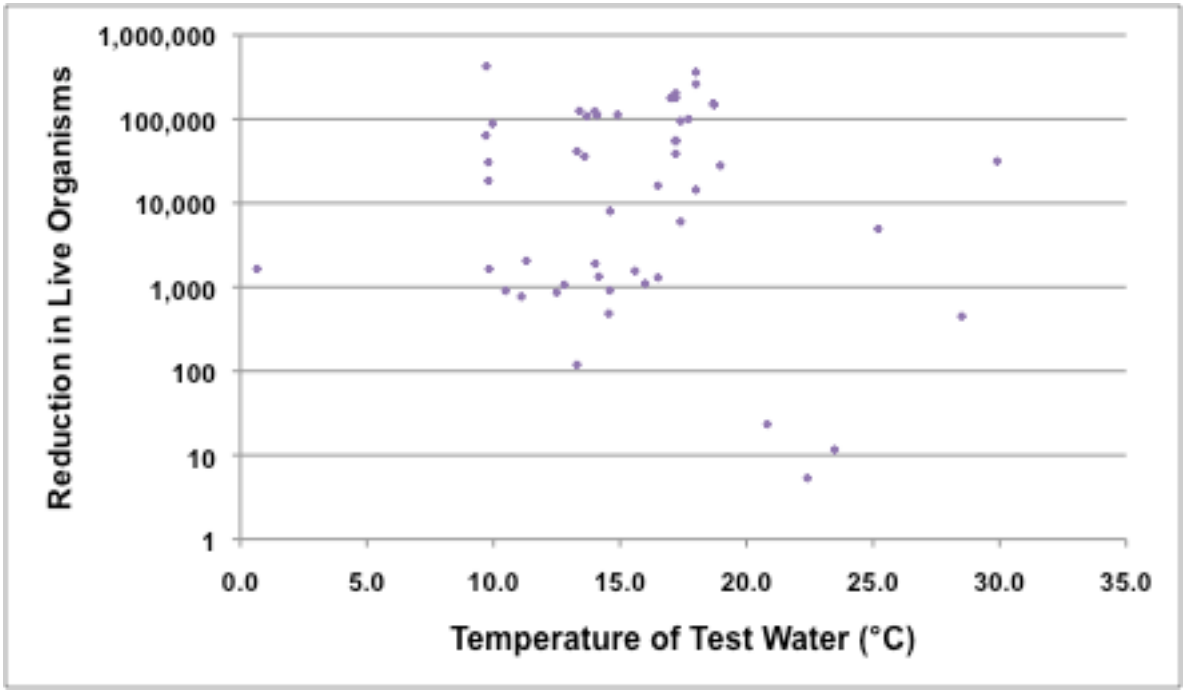


Figure A-10. Zooplankton Reduction versus Temperature: Brackish Water Trials (1-20 psu). y=-1110x+90873, n=172, r2=0.0017, p<0.596.

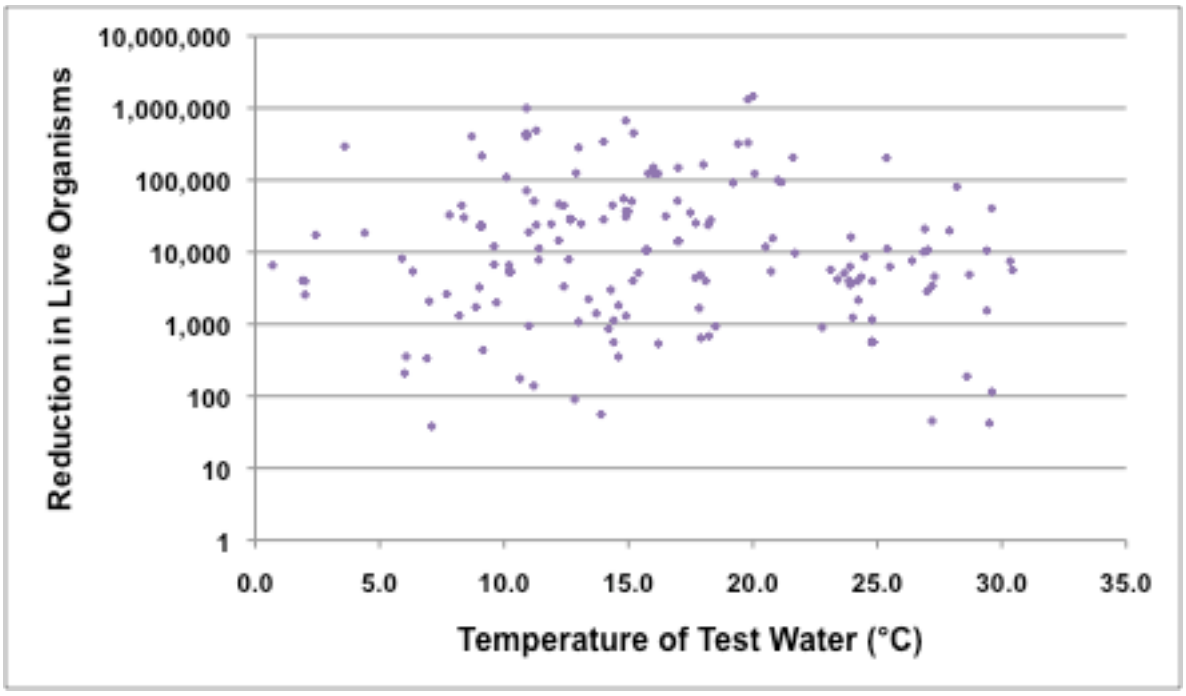
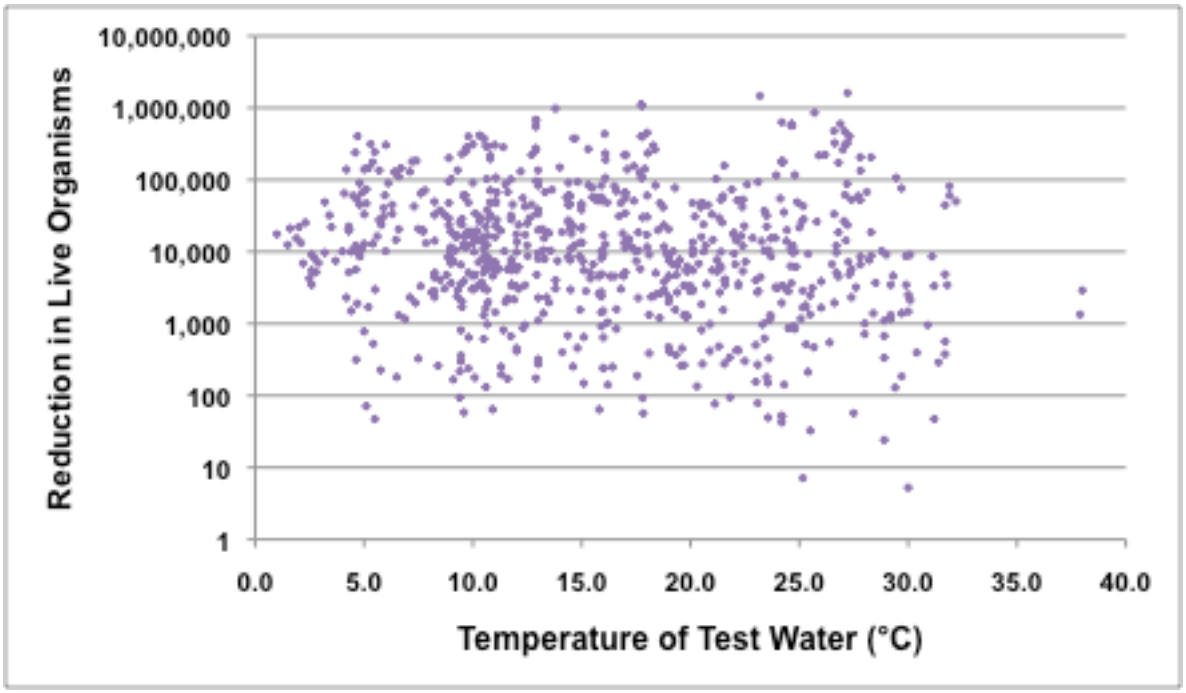


Figure A-11. Zooplankton Reduction versus Temperature: Saltwater Trials (>20 psu). y=1037x+41924, n=762, r2=0.0029, p<0.141.



(d) Holding Times

Figure A-12. Zooplankton Reduction versus Temperature: Trials with holding time < 5 days.
y=3327x+-7055, n=237, r2=0.0226, p<0.021.

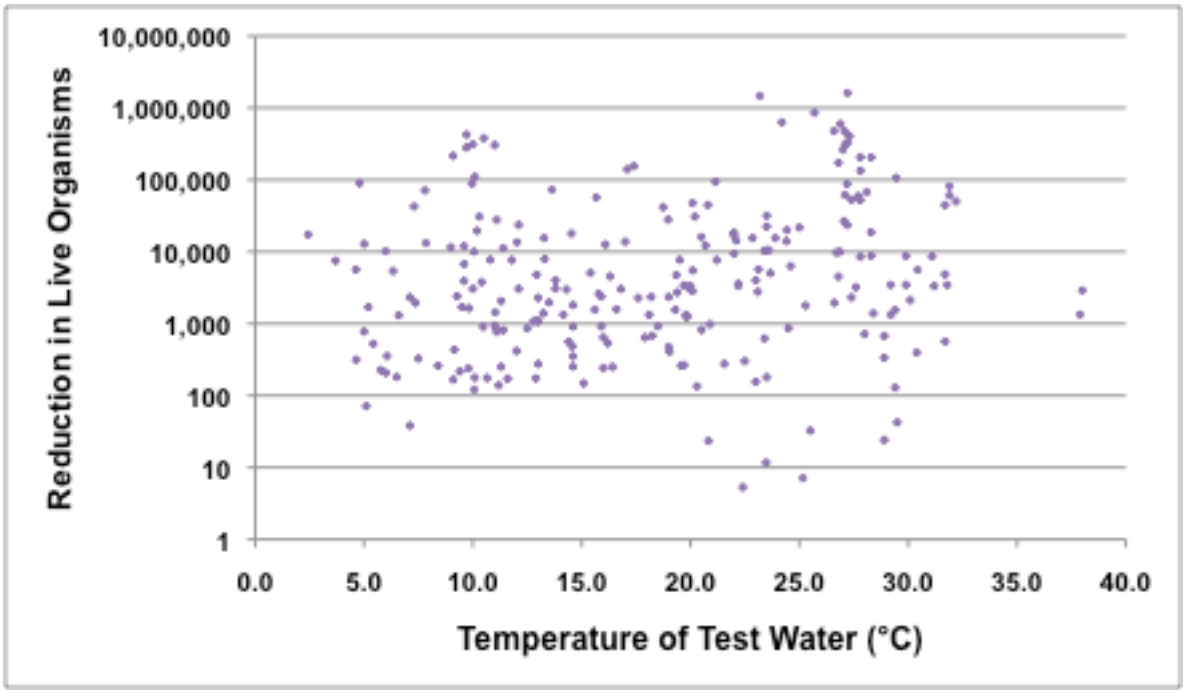


Figure A-13. Zooplankton Reduction versus Temperature: Trials with holding time = 5 days.
y=423x+63648, n=649, r2=0.0003, p<0.635.

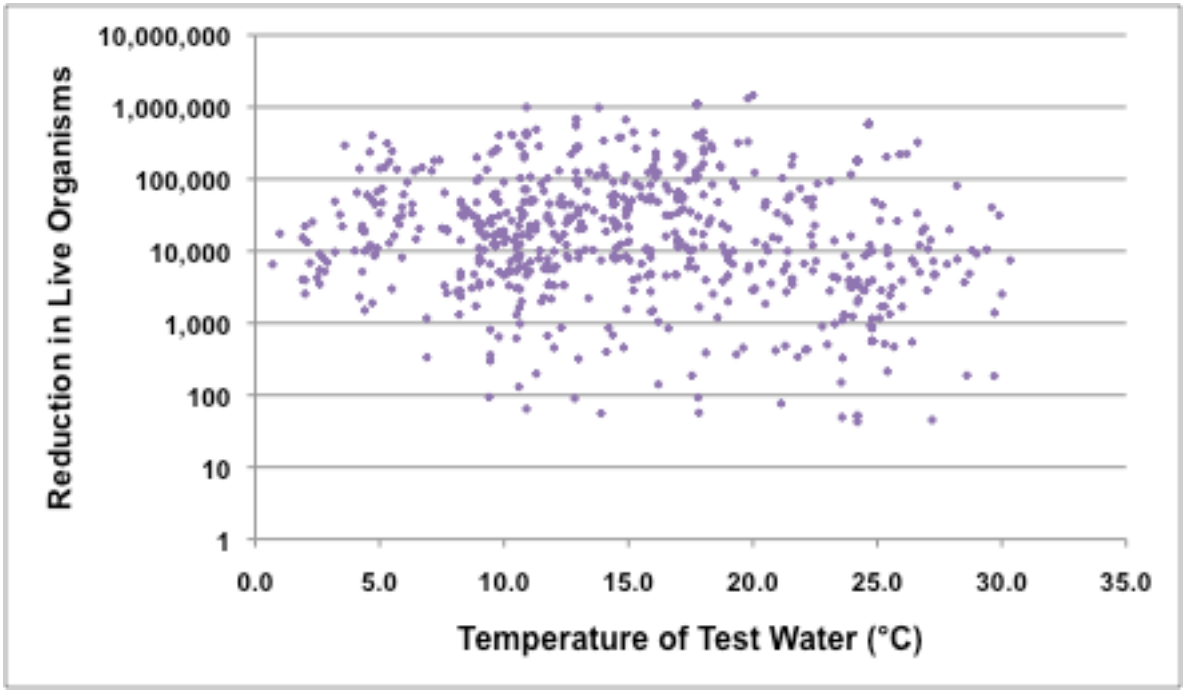
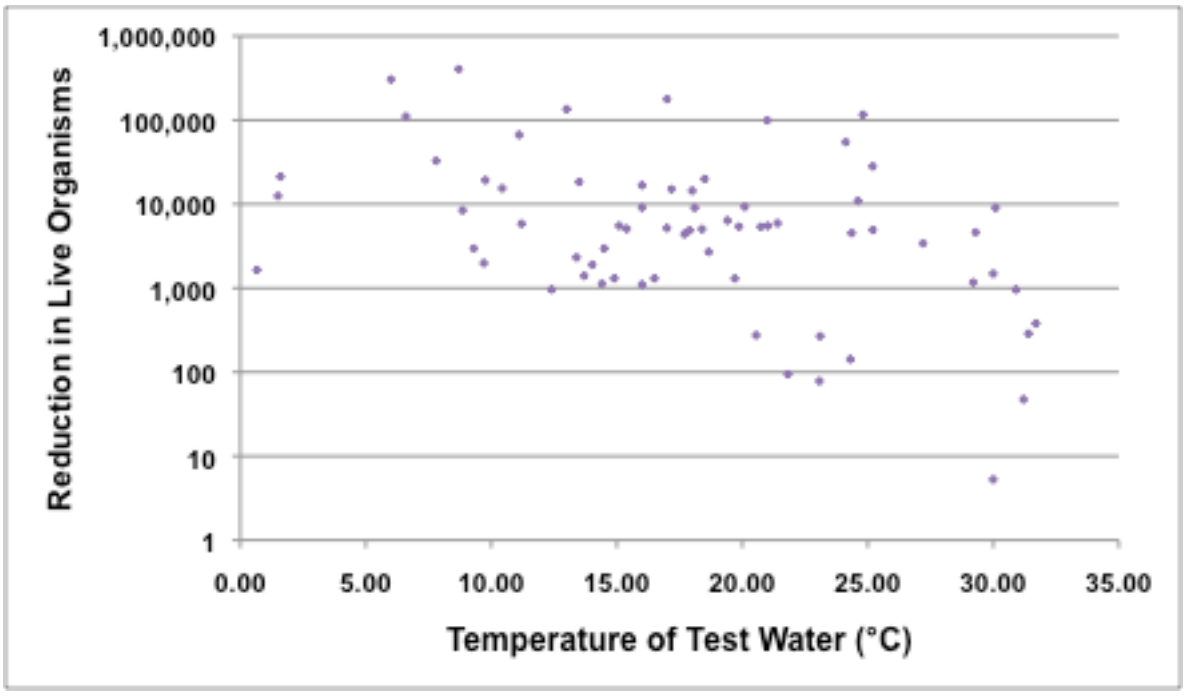
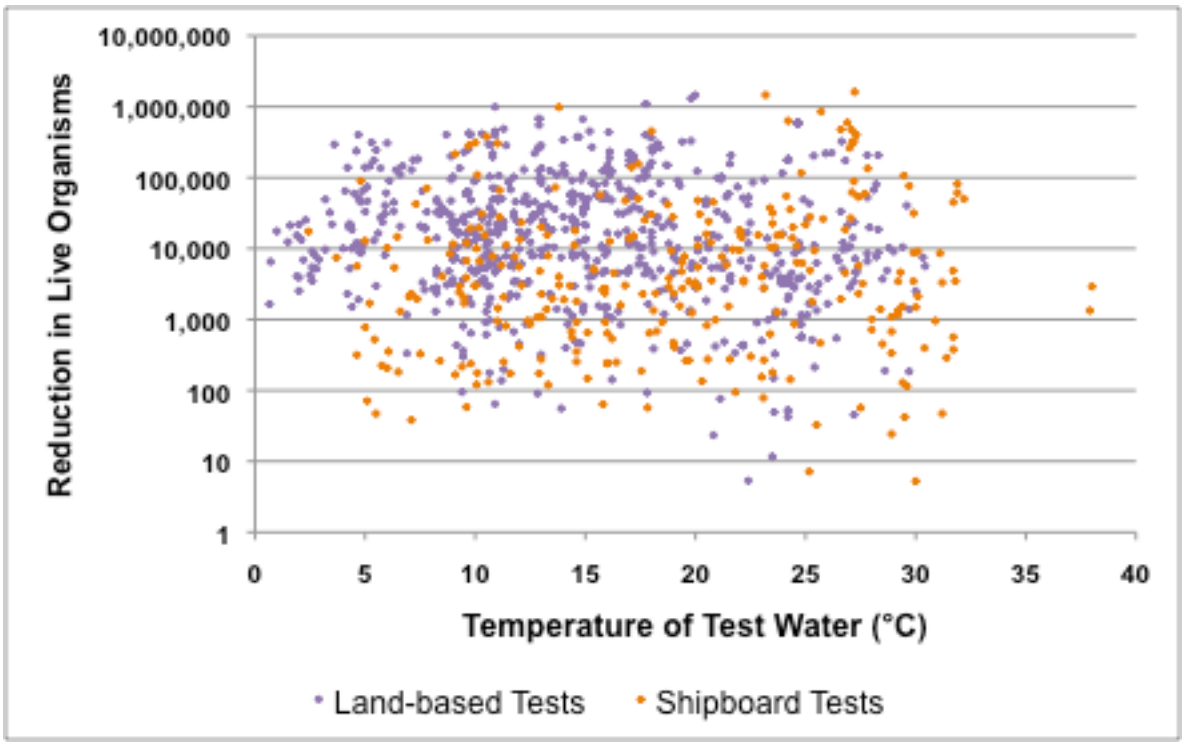


Figure A-14. Zooplankton Reduction versus Temperature: Trials with holding time > 5 days. y=-2460x+71579, n=68, r2=0.0768, p<0.022.



(e) Type of Trial

Figure A-15. Zooplankton Reduction versus Temperature: Land-based and Shipboard Trials. Land-based: y=238x+63495, n=696, r2=0.0001, p<0.768. Shipboard: y=2323x+4466, n=289, r2=0.0115, p<0.069.



2. Distribution among temperature brackets of publicly available data on treatment system performance and water temperature

Figure A-16. Zooplankton data: Number of trials per temperature bracket for AMS treatment systems.

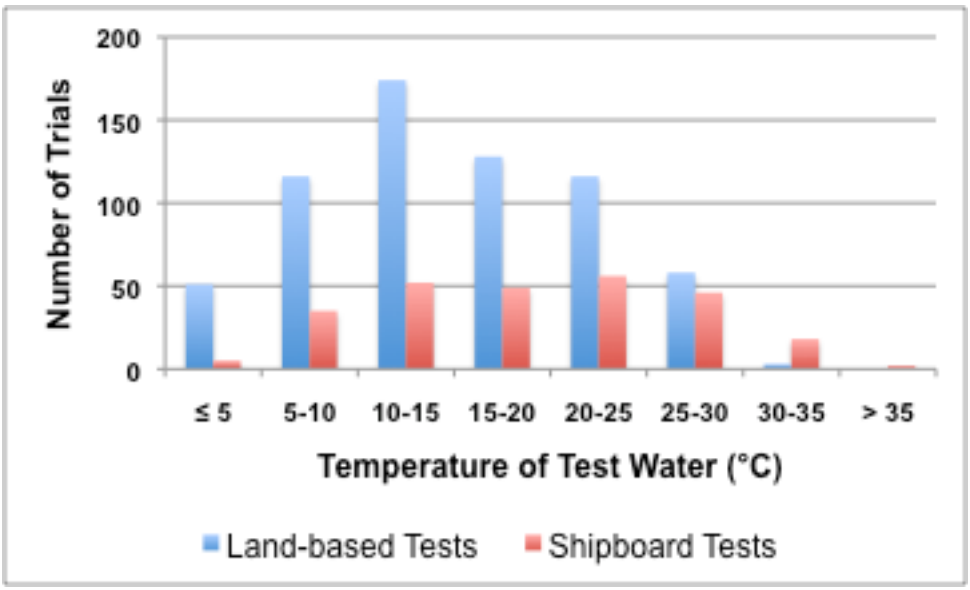


Figure A-17. Zooplankton data: Number of trials per temperature bracket for US type-approved treatment systems.

