

Early detection monitoring for larval dreissenid mussels: how much plankton sampling is enough?

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Abstract The development of quagga and zebra mussel (dreissenids) monitoring programs in the Pacific Northwest provides a unique opportunity to evaluate a regional invasive species detection effort early in its development. Recent studies suggest that the ecological and economic costs of a dreissenid infestation in the Pacific Northwest of the USA would be significant. Consequently, efforts are underway to monitor for the presence of dreissenids. However, assessments of whether these efforts provide for early detection are lacking. We use information collected from 2012 to 2014 to characterize the development of larval dreissenid monitoring programs in the states of Idaho, Montana, Oregon, and Washington in the context of introduction and establishment risk. We also estimate the effort needed for high-probability detection of rare planktonic taxa in four Columbia and Snake River reservoirs and assess whether the current level of effort provides for early detection. We found that the effort expended to monitor for dreissenid mussels increased substantially from 2012 to 2014, that efforts were distributed across risk categories ranging from high to very low, and that substantial

gaps in our knowledge of both introduction and establishment risk exist. The estimated volume of filtered water required to fully census planktonic taxa or to provide high-probability detection of rare taxa was high for the four reservoirs examined. We conclude that the current level of effort expended does not provide for high-probability detection of larval dreissenids or other planktonic taxa when they are rare in these reservoirs. We discuss options to improve early detection capabilities.

Keywords Early detection · Monitoring · Invasive species · Quagga mussel · Zebra mussel · *Dreissena polymorpha* · *Dreissena rostriformis bugensis* · Columbia River · Snake River

Introduction

The goal of early detection monitoring programs for invasive species is to detect new arrivals soon after their introduction so that eradication and quarantine measures can be used to control their spread (Mehta et al. 2007). Unfortunately, detection of newly introduced invasive species is often not possible until populations are well established because the effort expended is insufficient to detect them at low densities (Barry 2004; Hayes et al. 2005; Lodge et al. 2006; MacIsaac et al. 2002). Once invasive species populations achieve high densities, control methods become more costly or ineffective and the probability that the established population has become a propagule source for other areas will increase

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(Hulme 2006; Lodge et al. 2006). Consequently, managers are faced with the challenge of balancing the costs of surveys for small populations versus the costs of broader-scale eradication if monitoring fails to detect a population in the initial stages of invasion.

The capacity of invasive species monitoring programs to detect species early after introduction is often not known or considered. The discovery of introduced species is related to sampling procedures (Costello and Solow 2003), including collection procedures (Rees et al. 2014), the level of effort expended (Hoffman et al. 2011), and the distribution of efforts across the landscape (Inglis et al. 2006). Hoffman et al. (2011) used species accumulation theory to describe the probability of detecting rare species and suggested that the problem of detecting a newly introduced species is analogous to that of detecting a rare one. To quantify the effort required for high-probability detection of nonnative species, Hoffman et al. (2011) used a nonparametric method to estimate the number of samples required to obtain a near-total (95%) or total (100%) census of an asymptotic sample-based species richness estimator (Chao et al. 2009) for zooplankton, benthic invertebrates, and fish. Hoffman et al. (2011) estimated the effort required to detect 95% or more of species present in Duluth-Superior harbor could exceed 750 zooplankton samples, 150 benthic invertebrate samples, and 100 fish samples. However, such quantitative assessments of the efficacy of invasive species monitoring programs are rare.

The threat of an impending invasion coupled with the knowledge that an infestation will require significant mitigation to reduce damages often results in efforts to detect the invader. The states of Idaho, Montana, Oregon, and Washington (hereafter referred to as the Pacific Northwest) and Wyoming are the last remaining states in the contiguous USA where quagga (*Dreissena rostriformis bugensis*) and zebra (*Dreissena polymorpha*) mussels (hereafter referred to as dreissenids) have not been documented as being established (Wells et al. 2011). Numerous studies have examined the economic costs of mitigating the effects of dreissenid infestations that include a variety of deleterious effects on industrial and municipal water delivery systems and hydroelectric facilities (Connelly et al. 2007; IEAB 2010; IEAB 2013; MacIsaac 1996; Park and Hushak 1999; Prescott et al. 2013; Robinson et al. 2013). Recent studies examining the potential costs of a dreissenid infestation in the Columbia River (CR) Basin have suggested that the economic costs will be significant. The Independent

Economic Advisory Board (IEAB) estimated costs that include a combination of hydropower production losses, cleaning and control costs, costs of redundant screens, and new bypass systems at Snake River (SR) hydroelectric dams alone to be in the 100s of millions of dollars, with unknown additional costs associated with losses to fish hatcheries and ecosystem-level effects (IEAB 2010). The deleterious effects of dreissenids on ecological function, native flora and fauna, and water quality have also been studied and reported in areas with known infestations (Higgins and Zanden 2010; Roper et al. 1995; Tatem and Theriot 1994). Consequently, the establishment of dreissenids could also affect mitigation efforts to conserve, protect, and restore important natural resources in the CR Basin, including those directed at recovering endangered and threatened Pacific salmon species (*Oncorhynchus* spp.). Multiple invasions of other invertebrate species into the CR have been documented, but the organisms were well established when they were discovered (Bollens et al. 2012; Dexter et al. 2015; Emerson et al. 2015; Hassett et al. 2016). In response to the potential ecological and economic costs of a dreissenid infestation and the presence of increasing introduction vectors (IEAB 2013; ISDA 2012; ODFW 2012), dreissenid monitoring programs have been established throughout the Pacific Northwest.

The formation of dreissenid mussel monitoring programs in the Pacific Northwest provides a unique opportunity to evaluate a regional invasive species detection effort early in its development. Despite the significant effort being expended to develop and conduct detection monitoring programs in the Pacific Northwest, little is known about the probability of detecting newly established populations. Understanding the effort needed to detect dreissenids when they are rare will allow managers to discern whether the level of effort currently being expended is sufficient to detect newly established populations early enough to enact control and quarantine measures (Heimowitz and Phillips 2011). Our goal is to provide an objective assessment of dreissenid monitoring programs in the Pacific Northwest. To accomplish this goal, we provide a summary of the recent (2012–2014) effort being expended to monitor for the presence of larval dreissenids and place these efforts in the context of estimates of introduction and establishment risk. We also provide estimates of the effort necessary to have a high probability of detecting dreissenids, and other introduced planktonic taxa, in four Columbia and Snake River reservoirs. We then assess expectations for early

detection given the current monitoring effort being expended in these areas.

Materials and methods

Study area

The Pacific Northwest is bounded by the Pacific Ocean to the west and the Rocky Mountains to the east, and contains part of the Cascade Mountain range (Fig. 1). The Columbia River Basin, which includes the Columbia and Snake rivers, encompasses a substantial portion of the Pacific Northwest (Fig. 1). The CR is one of the largest river systems in North America and flows through multiple jurisdictions. The CR originates in British Columbia, Canada; approximately 15% of the approximate 672,000 km² of the CR Basin is located in

Canada. The Columbia and Snake rivers contain multiple hydroelectric projects that are subject to multiple international treaties and other national legislation that govern aspects of water usage. The CR provides benefits to multiple user groups that include habitat for fish and wildlife, including some species listed as endangered or threatened under the Endangered Species Act, fisheries for tribes, water for irrigation of agricultural crops, drinking water for municipalities, power generation by hydroelectric facilities, and shipping and barge navigation. The SR is the 13th longest river in the USA and drains 280,000 km² in portions of Idaho, Nevada, Oregon, Utah, Washington, and Wyoming (Kammerer 1990). The SR is the largest tributary of the CR, comprising about 41% of the entire CR Basin. The average SR discharge at its confluence with the CR constitutes 31% of Columbia's average discharge at that point (USGS 2009). Plankton samples used in this study were

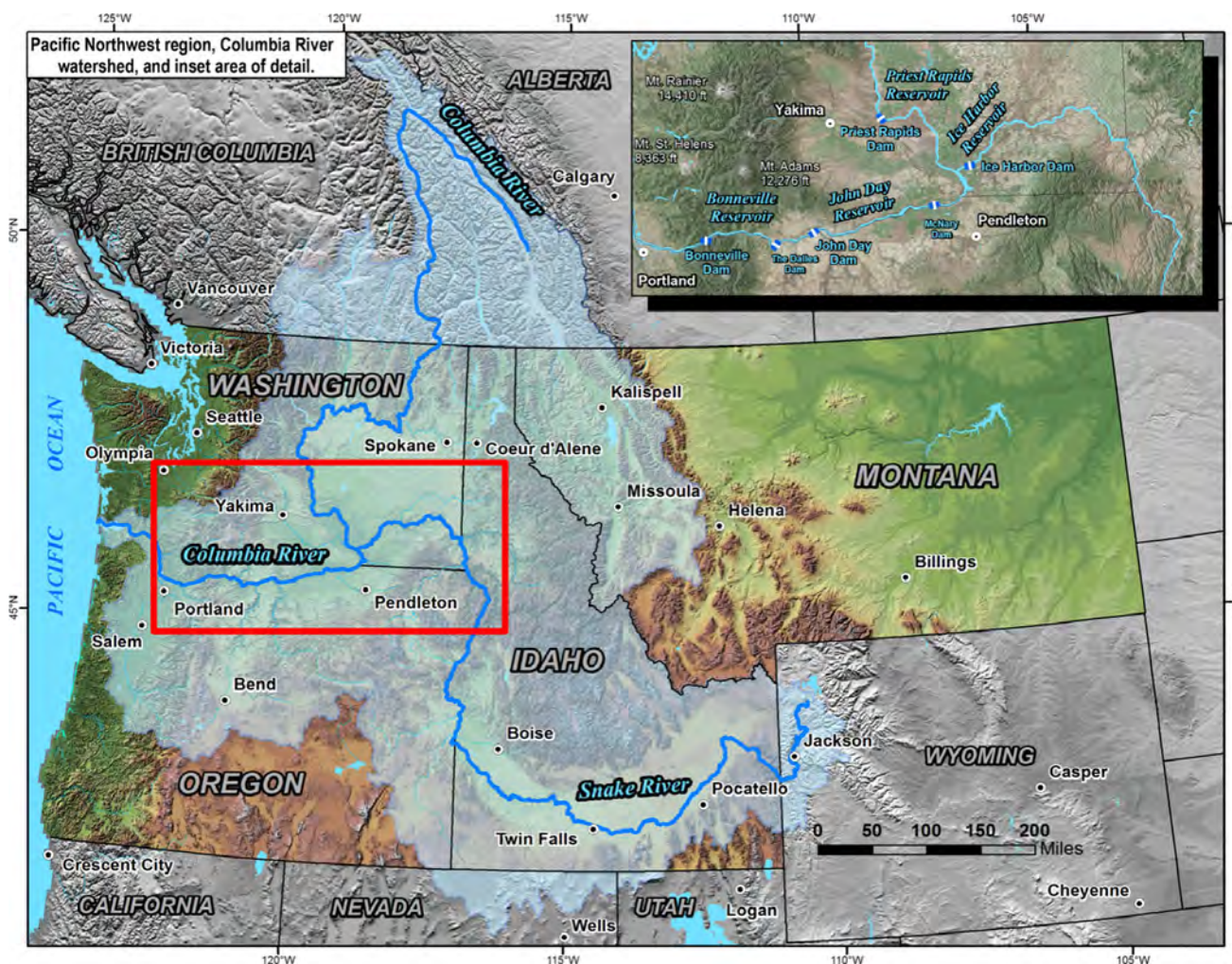


Fig. 1 Map of the study area that includes the states of Idaho, Montana, Oregon, and Washington. The Columbia and Snake rivers are the two largest rivers in the Columbia River Basin which is shown as a shaded area on the map

collected in four reservoirs: Bonneville, John Day, and Priest Rapids reservoirs on the CR and Ice Harbor Dam reservoir on the SR (Fig. 1).

Summarizing regional monitoring efforts for larval dreissenids

During 2013–2015, we queried regional entities conducting monitoring for the presence of dreissenids in the Pacific Northwest to send us the spatial extent of their monitoring activities and also data that would allow us to estimate the effort being expended. Requests for information were sent to a variety of different federal, private, state, and tribal entities that were known to be conducting monitoring (Table 1). We subsequently received georeferenced locations of the monitoring efforts and other associated data for the 2012, 2013, and 2014 monitoring seasons that typically began in April and ended in September.

To characterize the distribution of effort across risk of dreissenid mussel introduction and establishment categories, we summarized the number of plankton tows occurring by risk category and the state in which the monitoring was conducted. To assess the risk of introduction and establishment of waterbodies in the Pacific Northwest, we adopted the risk of introduction and establishment classifications put forth in Wells et al. (2011); with the exception of Montana, where we used Montana Fish, Wildlife, and Parks estimates of total angling pressure (<http://fwp.mt.gov/fishing/anglingData/anglingPressureSurveys/2013>.

[html](http://fwp.mt.gov/fishing/anglingData/anglingPressureSurveys/2013)) to estimate the risk of introduction. Wells et al. (2011) used a combination of expert judgment and data to formulate a list of waterbodies to evaluate for risk of introduction and establishment. Waterbodies were selected for evaluation if they were labeled on maps or were otherwise recognized for recreational use. Multiple sources were then queried to compile water quality and boater recreational data for these waterbodies. For water quality data, the U.S. Environmental Protection Agency (EPA) STORET database (<http://www.epa.gov/storet>) and the USGS National Water Information System (NWIS) database (<http://waterdata.usgs.gov>) were the primary sources. Other sources of water quality data included data from state agencies, peer-reviewed literature, and government agency reports (Wells et al. 2011). Risk of establishment was based on calcium concentrations (Table 2). Wells et al. (2011) based risk of introduction for each waterbody on quartiles of recreational boating use data (Table 3). Recreational boating use was determined from annual boating and angling pressure (i.e., use days, trips), angling tournaments, and state assessments of recreational use. The data describing recreational boater use was not consistent between states; therefore, in contrast to the risk of establishment classification, the risk of introduction categories was specific to each state. Total pressure (e.g., total use days) and the number of registered angling tournaments were the most commonly used parameters to quantify recreational boating. Similar to Wells et al. (2011), we grouped data describing total angling pressure in Montana into quartiles

Table 1 Agencies conducting monitoring for the presence of larval dreissenid mussels that participated in the survey and the state(s) they monitored

Agency	State(s) monitored
Chelan County Public Utility District	Washington
Douglas County Public Utility District	Washington
Grant County Public Utility District	Washington
Idaho Department of Environmental Quality	Idaho
Idaho State Department of Agriculture	Idaho
Libby Mitigation	Montana
Montana Fish, Wildlife, and Parks	Montana
National Park Service	Montana
Portland State University	Oregon, Washington
Spokane Tribe	Washington
US Army Corps of Engineers	Idaho, Montana, Oregon, Washington
US Bureau of Reclamation	Idaho, Montana, Oregon, Washington
US Geological Survey	Oregon, Washington
Washington Dept. of Fish and Wildlife	Washington
Washington State University, Vancouver	Oregon, Washington

Table 2 Values of dissolved calcium (mg/L) used to assign a risk category to individual waterbodies for determining the likelihood of dreissenid mussel establishment

Risk of establishment category	[Ca ²⁺] (mg/L)
High	>25
Medium	>15 and ≤25
Low	>12 and ≤15
Very low	≤12
Missing	No data

and then classified them as being of high, medium, low, or very low risk of introduction. For all classifications of risk of establishment and introduction, we then assigned these categorical classifications with numerical values such that high = 1.00, medium = 0.75, low = 0.50, and very low = 0.25. We then multiplied the estimated risks of establishment and introduction to form a composite index of risk. The waterbodies were then classified into the following categories based on the value of the composite index: ≥ 0.75 = high, < 0.75 and ≥ 0.50 = medium, < 0.50 and ≥ 0.25 = low, and < 0.25 = very low.

In addition to characterizing the monitoring efforts in the context of relative risk, we also queried the entities conducting the monitoring (Table 1) about the reasons why they selected their monitoring locations; we summarized their responses and present the results for the 2014 monitoring season. The query was included as a column in a spreadsheet that was sent to the agencies listed in Table 1 so that they could characterize multiple aspects of their monitoring activities (e.g., location, method, effort, date, etc.). The question posed was simply: What was the basis for choosing the locations you monitored for larval quagga/zebra mussels? The agencies were offered the following range of potential responses:

Table 3 The value ranges (Q = quartile) of recreational boater use data that were assigned to risk of introduction categories. Recreational boater use data were not consistent between states, and risk is assigned relative to waterbodies with data in each state

Risk of introduction category	Quartile ranges of recreational use data
High	$>Q_3$
Medium	$>Q_2$ and $\leq Q_3$
Low	$>Q_1$ and $\leq Q_2$
Very low	$\leq Q_1$
Missing	No data

(i) legacy: a site that has been visited for multiple years for detection monitoring, (ii) risk assessment: the waterbody indicated as high risk based on published risk assessments; (iii) water quality: unpublished calcium data indicate risk of establishment, (iv) recreational use: perceived to be at risk of introduction based on unpublished use data, and (v) opportunistic: site is visited for other studies and detection monitoring is conducted in addition to other sampling efforts. All of the agencies listed in Table 1 provided information. However, not all sites listed as being monitored were tagged with information. We did not follow up with the agencies as to the reasons why the response was missing. The responses from the agencies listed in Table 1 were then summed by state.

Estimating the effort needed for high-probability detection of rare planktonic taxa

We used data collected by Emerson et al. (2015) as part of a 2-year field study of mesozooplankton in Bonneville, John Day, and Priest Rapids reservoirs on the CR and in Ice Harbor reservoir on the SR to estimate parameters needed to quantify the effort necessary for detection of rare planktonic taxa using standard techniques for assessing plankton communities. Zooplankton samples were collected monthly from the downstream lower portions of each of the reservoirs, between July 2009 and June 2011, as part of research of the seasonal dynamics of zooplankton in the CR (see Emerson et al. 2015 for additional methodological details). All sampling sites were between 3 and 8 km upstream from their respective dams at the deepest point in the river channel cross section (Bonneville = 27 m, John Day = 55 m, Priest Rapids = 26 m, Ice Harbor = 36 m). Each sampling location was typically at or near the center of the river, except at Bonneville, where the deepest waters occur close to the north shore. Triplicate zooplankton samples were collected at each site via vertical haul of a 0.5-m-diameter, 73- μ m mesh zooplankton net from 0.5 m off the reservoir bottom to the surface. A flow meter in the mouth of each net determined the volume of water filtered. Samples were rinsed from the net and preserved with a 5% solution of buffered formalin. Aliquots from each preserved zooplankton sample were taken from a homogenized whole sample with a Hensen-Stempel pipette. Individuals were then identified to the lowest possible taxon (Edmondson 1959; Balcer et al. 1984; Thorp and Covich 2010; Cordell 2012) using a Nikon SMZ 1500 microscope (Emerson et al. 2015).

To quantify the effort required for high-probability detection of rare planktonic taxa (Hoffman et al. 2011), we used species accumulation theory to estimate species richness and characterize species accumulation. Since we were primarily interested in assessing the level of effort needed to detect larval dreissenids in the Columbia and Snake River reservoirs, we used data collected by Emerson et al. (2015) during April–September, 2010, to account for seasonal differences in the plankton community, to focus our analyses on periods when dreissenid mussels are likely to be planktonic, and to coincide with the timing of current larval dreissenid monitoring efforts in the four Columbia and Snake River reservoirs. Counts of taxa were not adjusted for volumetric subsampling.

We estimated the number of plankton samples and subsequently the volume of water filtered by plankton nets required to detect a range of proportions of a species richness estimator using a nonparametric method proposed by Chao et al. (2009) (Hoffman et al. 2011). We calculated sample-based rarefaction curves and estimated total species richness (S_{est}) and associated 95% confidence intervals using the EstimateS v9.10 software (Colwell 2006). Total species richness was estimated using a nonparametric, sample-based estimator (Chao 1987; Chao et al. 2009): $S_{\text{est}} = S_{\text{obs}} + (1 - 1/t)Q_1^2/(2Q_2)$, where S_{obs} is the number of species in the sample set, t is the number of samples, and Q_1 and Q_2 are the number of “uniques” (species that occur in only a single sample) and “duplicates” (species that occur in only two samples), respectively (Hoffman et al. 2011). If $Q_2 = 0$, then the formula is replaced by $S_{\text{est}} = S_{\text{obs}} + (1 - 1/t)Q_1(Q_1 - 1)/(2(Q_2 + 1))$ (Chao 2009). We then used methods described in Chao et al. (2009) to estimate the number of samples required to detect a variety of proportions of the asymptotic sample-based species richness estimator S_{est} (“Chao2”); we specifically report the volume of filtered water needed to detect 90, 95, 99, and 100% of S_{est} . We estimated the volume of filtered water needed to achieve a proportion of S_{est} by multiplying the estimated number of samples by the average volume of filtered water sampled per plankton tow for a particular reservoir (Emerson et al. 2015) during April–September 2010.

To assess the estimated proportion of taxa that we would expect to detect given the recent level of effort (volume of filtered water) expended in a particular reservoir, we used data from entities that conducted detection monitoring in Bonneville, John Day, and Priest Rapids reservoirs on the CR and in the Ice Harbor reservoir on the SR during 2014. Field methods associated with the

collection of plankton samples to monitor for the presence of larval dreissenids during 2014 were somewhat variable. However, plankton nets were typically 0.15 m in diameter (range = 0.15–0.5 m) with a 64- μm mesh. For our analyses, we tallied the volume of water filtered from plankton tows that were vertical, oblique, or horizontal and collected from shore, but not for efforts where plankton nets were towed behind boats. The protocols for efforts that involved towing plankton nets behind boats, including accounting for boat and water speed and/or net clogging, were poorly documented. Consequently it was difficult to assess the integrity of these efforts; thus we excluded these data from our analyses.

Results

The total effort expended to monitor for the presence of larval dreissenids increased from 2012 to 2014 in all of the states we evaluated. In Idaho, the effort expended increased from 550 to 681 plankton tows; in Montana, from 161 to 617; in Oregon, from 239 to 740; and in Washington, from 209 to 675 plankton tows. The distribution of these efforts across risk categories varied among states (Table 4). In Idaho, the monitoring efforts were distributed across a variety of risk categories but the largest numbers of efforts were in the high- and low-risk categories. Increases in effort from 2012 to 2014 in Idaho occurred primarily in the high- and low-risk categories (Table 4). The predominant reason given for selecting sites in Idaho was “risk assessment” (Fig. 2). The distribution of monitoring efforts across risk categories was less variable for Montana, with the efforts being concentrated in waterbodies with either high risk or where risk data were missing; increases in the total effort expended to monitor for the presence of dreissenids were primarily in these two risk categories. The primary reason given for site selection in Montana was that the sites were “legacy” sites. For Oregon, the efforts were predominantly in areas designated high risk, with significant increases in sampling occurring in this category (2012, $n = 84$; 2014, $n = 413$). The predominant reason given for site selection in Oregon was risk assessment followed by “recreational use” (Fig. 2). In Washington, similar to Montana, the efforts were primarily distributed across waterbodies that were categorized as high risk and those where risk assessment data were missing (Table 4). Increases in effort from 2012 to 2014 were in these two risk categories, with the largest increase occurring for waterbodies missing

Table 4 Number of plankton tows (frequency) conducted by state and combined risk of establishment and introduction categories during 2012, 2013, and 2014

State	Risk category	2012 frequency	2013 frequency	2014 frequency
Idaho	High	121	116	167
Idaho	Medium	51	49	56
Idaho	Low	176	177	258
Idaho	Very low	89	94	93
Idaho	Missing	113	111	107
Montana	High	62	165	254
Montana	Medium	8	4	20
Montana	Low	0	0	1
Montana	Very low	0	1	7
Montana	Missing	91	237	335
Oregon	High	84	183	413
Oregon	Medium	21	20	36
Oregon	Low	76	105	170
Oregon	Very low	21	74	61
Oregon	Missing	37	39	60
Washington	High	69	234	281
Washington	Medium	6	9	10
Washington	Low	0	3	7
Washington	Very low	26	6	19
Washington	Missing	108	209	358

risk assessment data (2012, $n = 108$; 2014, $n = 358$). The predominant reason given for site selection in Washington was recreational use (Fig. 2).

The plankton rarefaction curves for Bonneville, John Day, and Priest Rapids reservoirs on the CR and Ice Harbor reservoir on the SR did not reach an asymptote, indicating that the plankton assemblages were not fully sampled (Fig. 3). The rarefaction curves indicated that the relationship between the number of samples collected and taxa detected was not consistent across reservoirs. The estimated total species richness varied among reservoirs, ranging from 34 taxa in John Day reservoir to 58 taxa in Bonneville reservoir (Table 5).

Our analyses relating the volume of filtered water to the estimated proportion of taxa detected suggest that the level of effort needed for high-probability detection of rare planktonic taxa, including larval dreissenids, is much higher than is currently being expended in the four Columbia and Snake River reservoirs (Fig. 4). As expected, the relationships between the volume of water filtered and the estimated proportion of taxa detected indicate a nonlinear relationship where much more

effort is needed to detect rare taxa than for more abundant taxa. The volume of filtered water needed to detect a particular proportion of planktonic taxa varied among reservoirs. The expected proportion of taxa detected was generally low for all reservoirs given the level of effort expended during 2014.

For Bonneville reservoir, we estimated that the volume of filtered water needed to detect 90% of the planktonic taxa was 438 m³, to detect 95% of taxa was 616 m³, to detect 99% of taxa was 959 m³, and to fully census the taxa present was 1559 m³. The volume of filtered water collected from Bonneville reservoir to monitor for larval dreissenids during 2014 was 40 m³. The estimated proportion of taxa that we would expect to detect given this volume of filtered water was less than 0.60 (Fig. 4).

The estimated volume of filtered water needed to detect a particular proportion of taxa was less for John Day reservoir than Bonneville reservoir. The estimated volume of filtered water needed to detect 90% of the planktonic taxa was 267 m³, to detect 95% of taxa was 467 m³, to detect 99% of taxa was 927 m³, and to fully

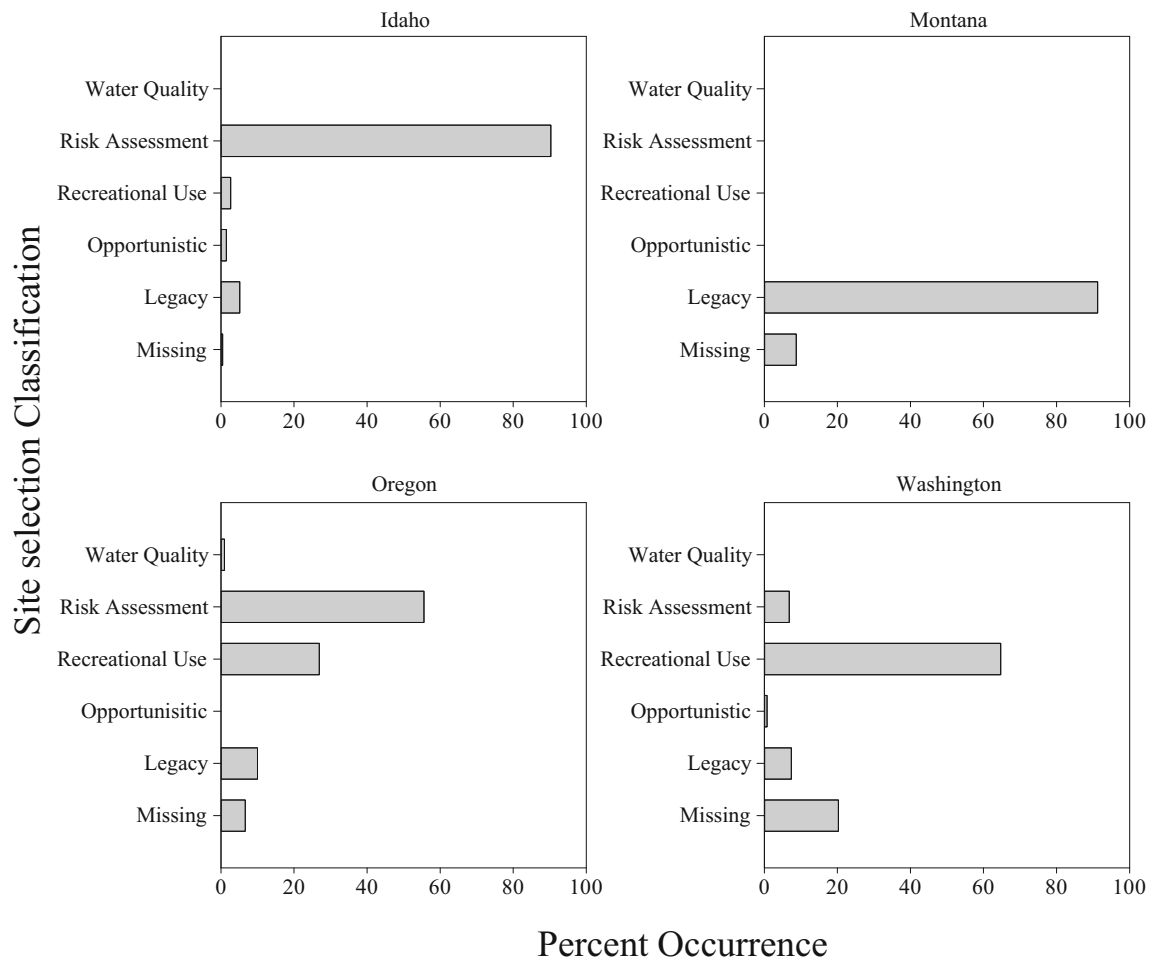


Fig. 2 The reasons for the selection of larval dreissenid mussel monitoring sites in the states of Idaho, Montana, Oregon, and Washington given by entities conducting the monitoring

census the planktonic taxa present was 1278 m³ (Fig. 4). The volume of filtered water collected from John Day reservoir during 2014 (101 m³) suggested that the expected proportion of taxa detected was less than 0.88.

The volume of filtered water needed to detect a particular proportion of taxa in Priest Rapids reservoir was lower than for both Bonneville and John Day reservoirs (Fig. 4): the volume needed to detect 90% of the planktonic taxa was 183 m³, to detect 95% of taxa was 292 m³, to detect 99% of taxa was 547 m³, and to fully census the taxa present was 690 m³. The volume of filtered water collected from Priest Rapids reservoir during 2014 (19 m³) was also much lower than for Bonneville and John Day reservoirs, indicating that the expected proportion of taxa detected was much less than 0.81 (Fig. 4).

For Ice Harbor reservoir, the last reservoir on the SR before its confluence with the CR, the volume of filtered water needed to detect a particular proportion of taxa was the highest among the four reservoirs examined

(Fig. 4). The volume of filtered water needed to detect 90% of the planktonic taxa was 564 m³, to detect 95% of taxa was 805 m³, to detect 99% of taxa was 1361 m³, and to fully census the taxa present was 1961 m³. The volume of filtered water collected from Ice Harbor reservoir during 2014 (84 m³) indicated that the expected proportion of taxa detected was less than 0.64 (Fig. 4). We conclude that the level of effort expended in these four Columbia and Snake River reservoirs was not sufficient to provide for early detection of larval dreissenid mussels.

Discussion

Efforts to monitor for the presence of quagga and zebra mussels in the Pacific Northwest, USA, are evolving in response to the threats posed by these invasive mussels. That sampling intensity increased from 2012 to 2014 is

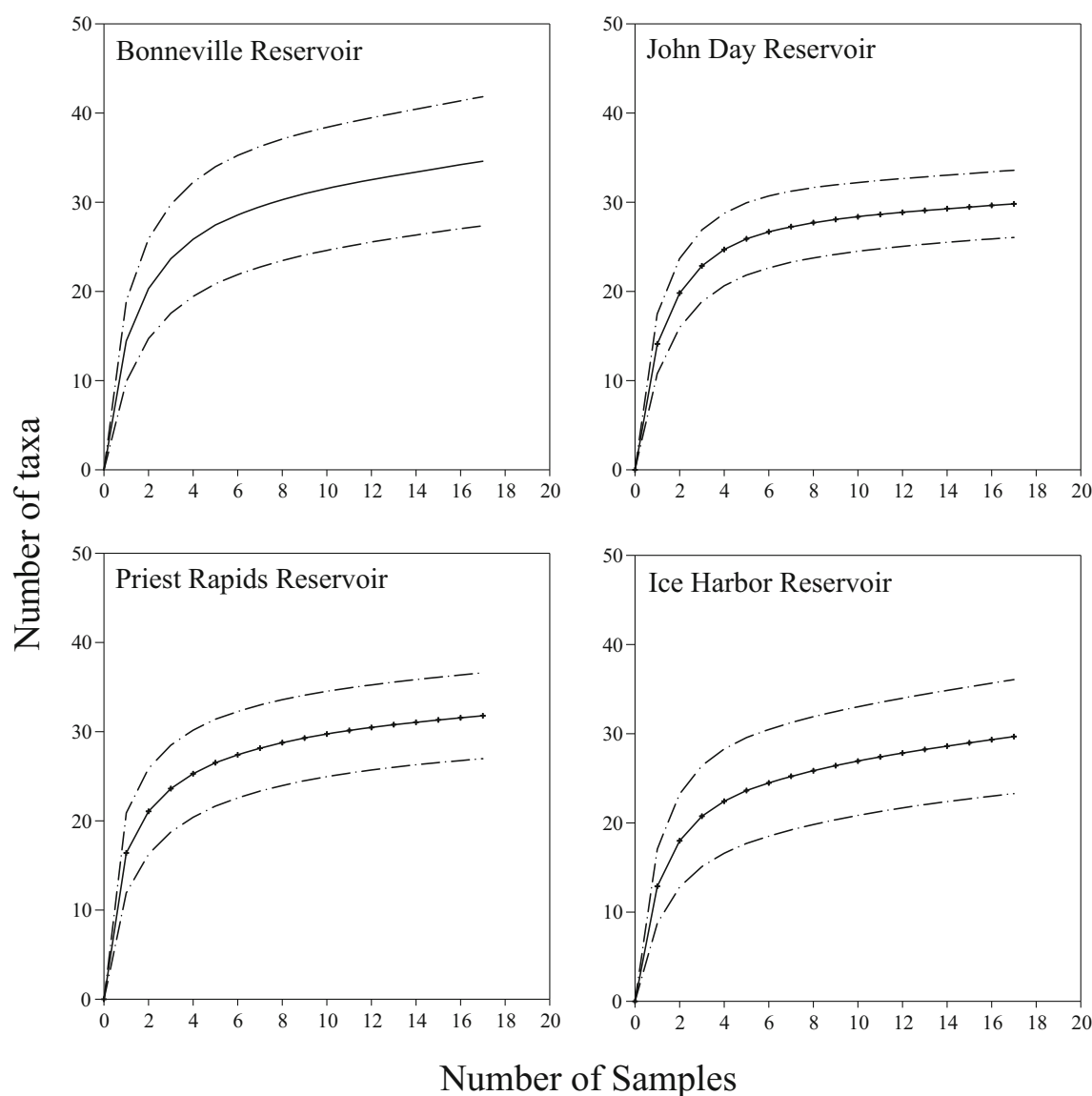


Fig. 3 Sample-based rarefaction curves for planktonic taxa collected by Emerson et al. (2015) during April–September 2010 in Bonneville, John Day, and Priest Rapids reservoirs on the

Columbia River and in Ice Harbor reservoir on the Snake River. Each plot shows the mean (solid line) with the associated 95% confidence intervals (dashed lines)

Table 5 The number of samples (n), number of uniques (Q_1), number of duplicates (Q_2), number of taxa observed (S_{obs}), the estimated number of planktonic taxa (S_{est}), and associated 95% confidence intervals (95% CI) based on samples collected by

Emerson et al. (2015) during April–September 2010, in Bonneville, John Day, and Priest Rapids reservoirs on the Columbia River and in Ice Harbor reservoir on the Snake River

River	Reservoir	n	Q_1	Q_2	S_{obs}	S_{est}	95% CI
Columbia	Bonneville	18	7	1	35	58	38–193
Columbia	John Day	18	3	1	30	34	30–69
Columbia	Priest Rapids	18	4	1	32	40	33–93
Snake	Ice Harbor	18	6	1	30	47	33–151

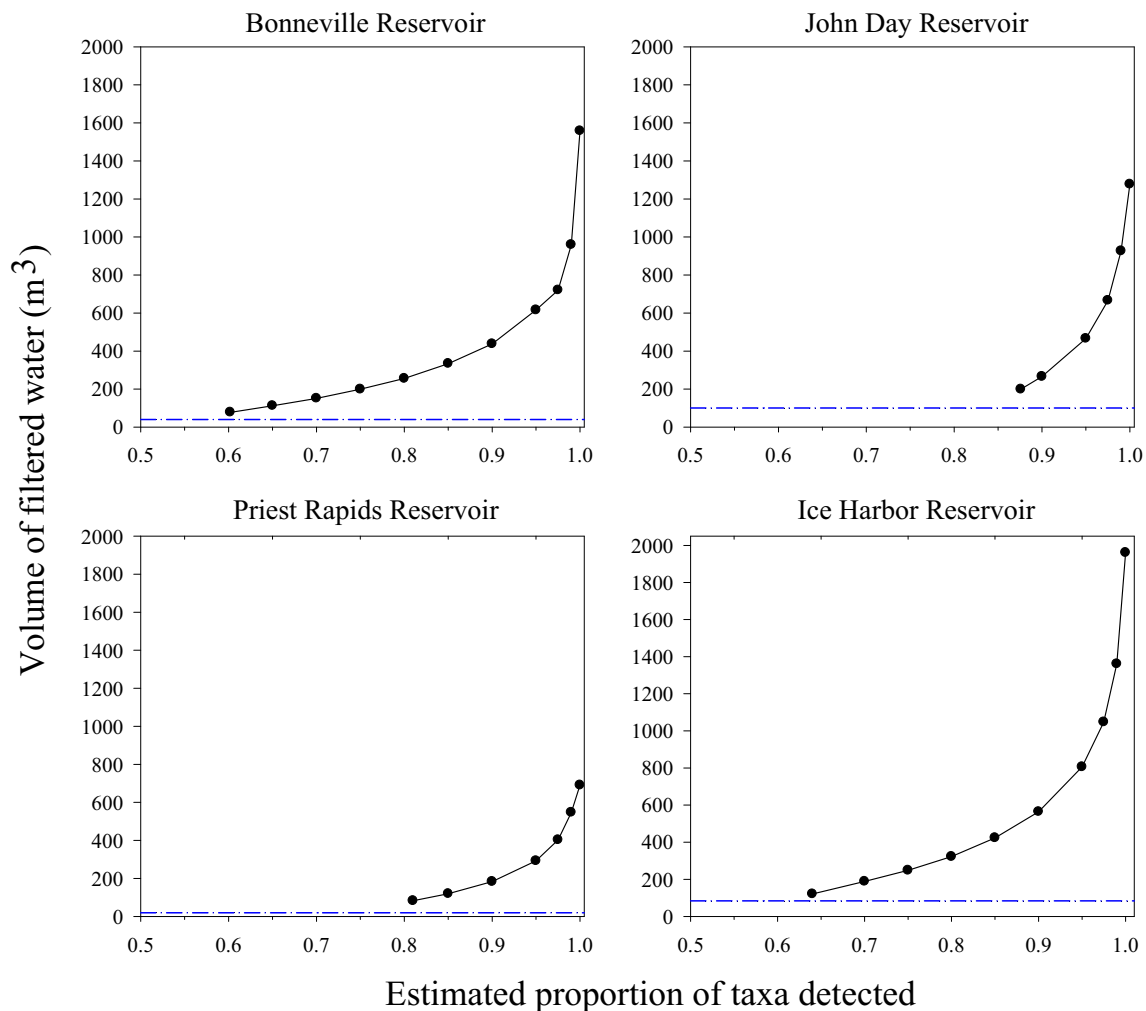


Fig. 4 Estimates of the proportion of plankton taxa detected for a given volume of water filtered (m^3) with plankton nets in Bonneville, John Day, and Priest Rapids reservoirs on the Columbia River and in Ice Harbor reservoir on the

Snake River. The volume of filtered water collected to monitor for larval dreissenid mussels during 2014 is shown as a blue dashed line

an expected and typical response to the threat of invasion by an organism that can cause harm to ecological and economic systems (Gotelli & Colwell 2001). Increases in monitoring sampling intensity have been called for in various forums (IEAB 2010), and the Pacific Northwest states have increased their efforts from 2012 to 2014 to monitor for the presence of dreissenids. Presumably, the increases in effort are intended to increase the probability of detecting dreissenids early enough to eradicate them or control their spread to other waterbodies. Eliminating infestations has clear benefits in that the threat is removed. However, merely delaying the spread of an infestation within and between waterbodies has benefits as well (Lee et al. 2007). Since the projected cost of an infestation of dreissenids in the Pacific Northwest is large

(IEAB 2010), delays in realizing the cost of control measures have economic benefits. Delaying the spread of dreissenids will also allow for the development and planning of response actions that could further limit the scope of an infestation and/or reduce management costs (IEAB 2010). In other areas of the USA, the risk of a dreissenid infestation has been shown to be related to the monitoring level, arrival prevention, and response planning (Lee et al. 2007). While we document increases in effort from 2012 to 2014, funding for monitoring continues to be unstable, despite the fact that prevention efforts have been demonstrated to be highly cost-effective in the long term (Leung et al. 2002; Lee et al. 2009). Finnoff et al. (2007) suggested that prevention efforts may be undervalued because of a perceived risk that invasion may occur even with prevention efforts.

Despite the increases in sampling intensity we observed from 2012 to 2014, the level of effort expended to detect the presence of dreissenids in the four Columbia and Snake River reservoirs is not likely to be sufficient for detecting larval dreissenids, or other planktonic organisms, when they are at low densities. Consequently, our analyses suggest that the current dreissenid monitoring programs in the Pacific Northwest are on the same trajectory as many previous monitoring activities for aquatic invasive species. That is, most discoveries of aquatic invasive species occur only after the organisms have established and achieved ecologically significant densities (Bax et al. 2002; Inglis et al. 2006; Myers et al. 2000). We provide estimates of the level of effort (i.e., volume of filtered water) needed to be able to detect rare planktonic taxa, but caution that efforts to increase the volume of filtered water need to account for gear limitations such as net clogging or other factors that would reduce gear effectiveness. Protocols exist that provide general guidelines for sampling for larval dreissenids with plankton nets. However, since factors that affect sampling efficiencies (e.g., plankton density) vary among waterbodies and seasonally, and little is known specifically about how these factors affect sampling efficiencies in waterbodies in the study area, more research is needed to provide meaningful guidelines. While the results of our analyses are not directly transferable to other waterbodies, our results suggest that the level of effort expended in most waterbodies in the Pacific Northwest may not be sufficient for early detection of dreissenids or even for documenting more abundant planktonic taxa. For example, based on data collected as part of this study, we found that 85% of the waterbodies sampled in the Pacific Northwest during 2014 had fewer than 10 plankton tows per waterbody and 64% had fewer than three plankton tows, while 56 plankton tows were conducted in the Bonneville reservoir.

In the absence of increased funding to collect and process additional samples, the options for increasing the efficacy of the current monitoring are few. Reallocating existing regional efforts to areas of higher risk, for example to areas where propagule pressure from areas already infested is highest and where dreissenids are most likely to become established and flourish, could increase the probability of detecting infestations within the Pacific Northwest (Harvey et al. 2009). Our results suggest that risk assessment data are being considered to some extent in the allocation of

monitoring efforts in the Pacific Northwest. However, while risk assessments for dreissenids have been conducted (Wells et al., 2011; Whittier et al. 2008), the consistent and effective application of these assessments is confounded by our incomplete knowledge of introduction and establishment risk. Not only is this true because of the incomplete spatial coverage of existing risk assessment data, but also because our understanding of the biological and ecological requirements of zebra and quagga mussels, of what allows dreissenids to become invasive, and of the nature and extent of propagule pressure is limited. With respect to the biological and ecological requirements of quagga and zebra mussels, there is existing literature to draw upon (Mackie and Schloesser 1996; McMahon and Ussary 1995; Mills et al. 1996; Nalepa and Schloesser 1992; Vanderploeg et al. 1996), but much uncertainty remains. In a recent workshop to define research and management priorities for dreissenids in the Pacific Northwest, a better understanding of what biotic and abiotic conditions limit distribution, growth, and fecundity of dreissenids, and whether zebra and quagga mussels differ in environmental preferences and tolerances, were ranked as high biological research priorities by experts from around the USA (Sytsma et al. 2015). Filling gaps in the spatial coverage of establishment risk assessment data and increasing our knowledge of what constitutes establishment risk for both quagga and zebra mussels will increase the utility of using establishment risk as a way to better allocate sampling effort.

Similarly, a better understanding of introduction risk will also facilitate the allocation of monitoring efforts to areas of high risk. Wells et al. (2011) compiled data from a variety of disparate sources to assess introduction risk of dreissenids in the Pacific Northwest, but concluded that a regional assessment of introduction risk was not possible given the available data. In addition to increasing the amount of effort to monitor for the presence of dreissenids, more effort is being expended to inspect trailered boats traveling to and within states that are not currently known to be infested with dreissenids (IEAB 2013). If information was collected on the origin and destination of these boats, this information could be used to develop an understanding of the nature and extent of propagule pressure; however, legal issues regarding the collection and distribution of this information have been noted (Giffin 2012). The formulation of statistical models that predict boater movements has also been cited as a research priority for the Pacific

Northwest (Sytsma et al. 2015). Gravity models have been developed for other regions (Leung et al. 2002) and have proven useful for modeling invasion pathways between noncontiguous locations.

The development of alternate detection technologies may also improve the efficiency of existing monitoring efforts and result in higher detection probabilities (Lodge et al. 2006). Since existing sample processing methods for plankton organisms are very labor intensive, alternate sample processing methods have been evaluated. For instance, optical imaging techniques for assessing plankton in samples are being evaluated and have the potential to increase detection capabilities by either reducing sample processing effort and therefore allowing more samples to be processed or allowing a larger proportion of a particular sample to be processed (Buskey and Hyatt 2006; Day et al. 2012; Hassett et al. 2016; Sieracki et al. 2010). Improved field detection methods for rare species, whether they be for detecting species that have become rare through population declines or to detect species that have been recently introduced, have been recognized. For instance, the rapid expansion of bighead carp and silver carp in the Mississippi River system has led to the development of environmental DNA (eDNA) detection methods (Jerde et al. 2013; Mahon et al. 2013). Rees et al. (2014) suggested that eDNA analysis can have considerable time and cost benefits over traditional survey methods, especially when surveying for rare species. Jerde et al. (2011) observed that it took 93 days of person effort to detect one silver carp by electrofishing, whereas 0.17 day was required when using eDNA. Egan et al. (2013) suggested that combining eDNA with light transmission spectroscopy may facilitate the rapid detection of invasive species in the field to lower the risk of the introduction or spread of harmful species.

Critical assessments of invasive species early detection programs are needed to avoid repeating the mistakes of previous programs that have failed to detect populations until they are well established. In this paper, we provide an example of a collaborative process that can be used to assess early detection programs for planktonic organisms in other regions. The data we used was the result of a collaborative effort by multiple jurisdictions to collect, document, and distribute information that facilitated these analyses. Given the predicted economic effects and potential for ecological disturbance from an infestation of dreissenids into the Pacific Northwest, it is important to communicate the status of efforts to monitor for the presence of dreissenids so that managers can make

informed decisions about how to allocate resources to this issue. With the implementation of boat inspection programs to reduce the propagule pressure of dreissenids in the Pacific Northwest, it is clear that boats with live specimens of dreissenids are entering the Pacific Northwest (IEAB 2013). Hoffman et al. (2011) noted that early detection survey design requires considerations that include assessments of the potential biases from the survey design and collection methods (Fitzpatrick et al. 2009), power to detect presence and estimate abundance of organisms (Harvey et al. 2009), and effort and cost associated with collection and taxonomic classification (Lodge et al. 2006). Using information collected collaboratively with entities actively involved with monitoring for the presence of dreissenids in the Pacific Northwest, we provide insight into these considerations. Given this information, managers should consider the trade-offs between monitoring level, arrival prevention, and response planning. Assessing the optimal allocation of resources among various strategies to cope with an infestation of an invasive species has been done in other areas. For example, Leung et al. (2002) employed a quantitative bioeconomic model to identify the optimal allocation of resources to prevention versus control. Leung et al. (2002) suggested that these types of analyses can help to facilitate the interaction between risk assessors and managers by providing a quantitative rationale for policy decisions. For the Pacific Northwest, actions that managers may consider include i) enhanced levels of dreissenid monitoring, ii) research into improved methods of sample collection and processing, iii) better assessments of both risk of introduction and risk of establishment, and iv) improved modeling of resource allocation to guide policy and management decisions.

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