The economic risk of *Elodea spp.* to commercial sockeye salmon fisheries and recreational floatplane pilots in Alaska

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By

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Abstract

The presented research applies Monte Carlo simulation to conduct a formal bioeconomic risk and decision analysis on elodea, an invasive aquatic plant threatening ecosystem services in Alaska. The approach melds a metapopulation model with local market and non-market economic valuation to show that upfront management of existing invasions is the optimal strategy minimizing long-term damages. The analysis accounts for social-ecological feedback, region-specific risk, and allows for integration of species distribution models to achieve higher spatial resolution. Without intervention, damages to commercial sockeye fisheries and recreational floatplane pilots would amount to a median loss in natural capital of $1.4 billion in 2015 USD (90% CI: $0.1, $9.4 billion), providing a lower bound to potential damages. Even though the range of uncertainty is large, the certainty of long-term damages requires investments targeted at eradicating current invasions and preventing new arrivals. The study serves as a critical first step towards risk management aimed at protecting productive ecosystems of national and global significance.

Table of Contents

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1 Introduction

1.1 Overview

Invasive species are an increasing threat to the health of aquatic ecosystems worldwide. Biological invasions are reducing the ecosystem services of industries dependent on productive ecosystems such as recreation and fisheries (Nunes and van den Bergh 2004; Rothlisberger et al. 2012). In the U.S., damages of existing invasions are estimated to amount to $120 billion annually (Pimentel 2005). While most research focuses on estimating damages of existing invasions, little research has taken a forward looking approach by predicting future impacts (Lodge et al. 2016; Jeschke et al. 2014). While estimating financial damages of existing invasions can lead to more public awareness, these studies are less relevant for management because they don’t inform about the future consequences of decisions. Most importantly, they are unable to inform decision-makers about the value of prevention. Thus, lack of damage forecasting can result in inadequate human response to protecting the most valuable ecosystems and can lead to waste of money and resources (Doelle 2003). For example, investments to reduce damages in already impaired ecosystems have likely lower social returns compared to investments preventing invasions in pristine ecosystems (David Finnoff et al. 2007).

For economic impact assessments to be more relevant to management agencies, comprehensive bioeconomic risk analysis can guide strategic management decisions (Lodge et al. 2016). Critical components of such analyses include forecasting costs and benefits over an ecologically relevant time period that captures the potential population dynamics of the invader and related changes to market and non-market values (J.F. Shogren et al. 2006). Furthermore, in cases where the invader is dispersed by humans, bioeconomic models can better predict risk if they account for landscape-wide spread (Epanchin-Niell and Hastings 2010). Since intervention can alter the spread of the invader, the linked social-ecological model accounts for these important feedback mechanisms and enables the evaluation of management decisions across the landscape (D. Finnoff et al. 2010; Holmes et al. 2010). Few studies have integrated all of the above characteristics to guide current resource management decisions often limited by the availability of local data (Maguire 2004; Lodge et al. 2016). Some have tried to overcome the lack of location-specific economic values by using less sophisticated approaches such as benefit transfer techniques (Holmes et al. 2010).

This research uses a novel bioeconomic approach combining ecological modeling of the human-driven spread of an invasive species with economic valuation of ecosystem services that are at risk. In addition, different social science techniques are used to elicit, quantify, analyse, and validate structured expert knowledge about the uncertain effects of an invasive species on fisheries. The approach is spatially explicit, accounts for uncertainty, and forecasts region-specific damages to multiple stakeholders over a 100-year period. The effects of management action on the landscape-wide spread of the invader are considered. Optimization is used to find strategic management alternatives that minimize future damages and weigh management costs against avoided damages.

1.2 Elodea ecology, management, and history in Alaska

Elodea spp. (elodea) is Alaska's first known submerged freshwater invasive plant and is considered a threat to the state's freshwater resources with wide ranging ecological and economic effects (J. M. Morton et al. 2014). Elodea reduces biodiversity, compromises water quality, affects dissolved oxygen levels, and changes the structure of aquatic vegetation affecting the trophic interactions between fish and macroinvertebrates (Barko and James 1998; Jeppesen et al. 1998; Burks, Jeppesen, and Lodge 2001; S Diehl and Kornijow 1998). The presence of elodea in salmon bearing streams and lakes can reduce the quality of spawning and rearing habitat (Groves et al., 2004; Merz et al., 2008). While the threats imposed by elodea on Alaska's salmon resources seem obvious, there is little known about how far and how fast elodea can spread into local salmon bearing streams and waterbodies and what effect it will have on salmon reproduction. The plant can also form dense mats clogging waterways and interfere with water-based recreation and transportation (Halstead et al. 2003; Johnstone, Coffey, and Howard-Williams 1985). In Alaska, it has impeded boat navigation and recreation (Friedman 2015) and is a concern for floatplane operation safety (Hollander 2015b; CH2MHILL 2005). Similar invasive aquatic plants have reduced lake front property values in other U.S. states between 16% and 19% (Horsch and Lewis 2008; Zhang and Boyle 2010; Olden and Tamayo 2014).

There are five species of Elodea. *Elodea canadensis* (Canadian waterweed) is native to North America between 35˚ and 55˚N and *E. nuttallii* (Nuttall's waterweed) roughly overlaps this range. *Elodea bifoliata* occurs in western North America and *Elodea potamogeton* and *E. callitrichoides* are native to South America (Cook and Urmi-König 1985). The plant prefers sand and small gravel substrate with large amounts of available iron; cold, static, or slow-moving water ; depth of up to nine meters; and water of low turbidity (Riis and Biggs 2003; Rørslett, Berge, and Johansen 1986). Since *Elodea* is known to be a nutrient scavenger, eutrophic waters are more supportive of heavy long-term infestations (Rørslett, Berge, and Johansen 1986; Mjelde et al. 2012). Elodea reproduction is primarily vegetative with stem fragments and vegetative buds rooting in new locations. Vegetative buds can survive desiccation, low temperatures, and being frozen in ice (Bowmer, Jacobs, and Sainty 1995). Elodea has some of the highest fragmentation and regeneration rates among aquatic invasive plants causing rapid dispersal and severe challenges for mechanical removal (Redekop, Hofstra, and Hussner 2016).

Figure 1.1 *Elodea spp*. in Alexander Lake, Alaska, June 2016.
Source: Heather Stewart, DNR

Elodea is tolerant of a wide range of environmental conditions and has successfully invaded aquatic ecosystems worldwide. *Elodea canadensis* and *E. nuttallii* aggressively invaded the British Isles in the 19th and early 20th century (Simpson 1984). Elodea is established in much of Europe with populations generally on the decline but high rates of invasion remain in northern Europe, Asia, Africa, Australia, and New Zealand (Josefsson 2011). Common human-related pathways of introduction include the aquarium trade, boats, and floatplanes (Sinnott 2013;

Strecker, Campbell, and Olden 2011; Johnstone, Coffey, and Howard-Williams 1985). Natural long-distance dispersal pathways include flooding as well as waterfowl and wildlife transport (Spicer and Catling 1988; Champion, Winton, and Clayton 2014).

Possible management actions include draining and drying, herbicides, the introduction of herbaceous fish, and mechanical removal through suction dredging or hand pulling for example to name a few (Josefsson 2011; Beattie et al. 2011). Fluridone and Diquat are herbicides known to be the most effective management options, while mechanical methods such as cutting, or suction dredging result in plant fragments causing populations to spread to new areas (Josefsson 2011). Fluridone is a systemic herbicide that is absorbed through plant shoots and disrupts photosynthesis. It has successfully been used to manage elodea in Alaska and other locations in the U.S.. At very low concentrations Fluridone selectively removes elodea with few non-target effects (Hamelink et al. 1986; Kamarianos et al. 1989; Schneider 2000). Diquat is a contact herbicide that is absorbed by the plant’s leaves where it interferes with respiration. It is slightly toxic to fish with no shown bioconcentration (Cochran 1994; Davies and Seaman 1968; Harper, Chisholm, and Chandrasena 2007). Diquat is commonly used in combination with Fluridone as a cost-effective method of preventing the spread from partial-lake to full-lake infestations.

Figure 1.2 Elodea in Lake Hood, Anchorage, 2015. The orange colored aquatic weed harvester can be seen in the top right. In the past, been used to harvest dense aquatic vegetation, to improve the safety of floatplane operations. Source: Heather Stewart, DNR

In Alaska, elodea was discovered in Chena Slough, Fairbanks, Interior Alaska, in 2010, drawing attention to an already established but until then largely ignored population detected in Cordova, Southcentral Alaska, in 1982. New, previously unknown infestations were found in every year since 2010, including locations in Interior Alaska and Southcentral Alaska (Figure 1.3). In 2011, elodea was discovered in Sand Lake, Anchorage, where introduction likely occurred through an aquarium dump. Detection surveys conducted in 2012 found elodea in six remote waterbodies in the Cordova area, in two additional urban lakes in Anchorage, and three lakes on the Kenai Peninsula. The likely pathways in these locations are human-caused through aquarium dumps, boat, and floatplane traffic. Other natural distribution mechanisms include flooding as well as waterfowl and wildlife (Sytsma and Pennington 2015).

Realizing the continued spread across the state, stakeholders and land management agencies started to take action. In 2012 and 2013, a bill to establish a rapid response fund was introduced in the 27th and 28th Alaska but in both instances was not passed. In 2013, three years after its first discovery, elodea was manually removed in Chena Slough during a control trial using a suction dredge on 0.59 acres of the 55 acres infested at the time (Lane 2014). In the same year, a memorandum of understanding (MOU) was created between the Alaska Department of Natural Resources (DNR), designated to be the lead agency for managing freshwater aquatic plants, Alaska Department of Fish and Game (ADFG), and Alaska Department of Environmental Conservation (DEC). The MOU was aimed at more efficient permitting and the development of a statewide plan to eradicate elodea and coordinate interagency response. As a first step, elodea and four other invasive aquatic plants were added to a list of quarantined invasive plants (State of Alaska 2016). In Anchorage, many stakeholder meetings were held to deal with controversy over appropriate management action and lead agency responsibilities (Sinnott 2014).

In 2014, more evidence accumulated that floatplanes are distributing elodea from urban source locations to remote rural waterbodies with the discovery of elodea in Alexander Lake, Matanuska-Susitna Borough. Elodea was mainly growing in the approach path to a cabin owned by a floatplane pilot residing on Sand Lake in Anchorage (Hollander 2014). One year later, elodea was also found in Lake Hood, Anchorage, one of the world's largest seaplane bases (Figure 1.2). In the same year, elodea was also discovered in Totchaket Slough along the Tanana River, at least 90 river miles downriver from the largely unmanaged infestation in Chena Slough which was found in 2010 (Friedman 2015).

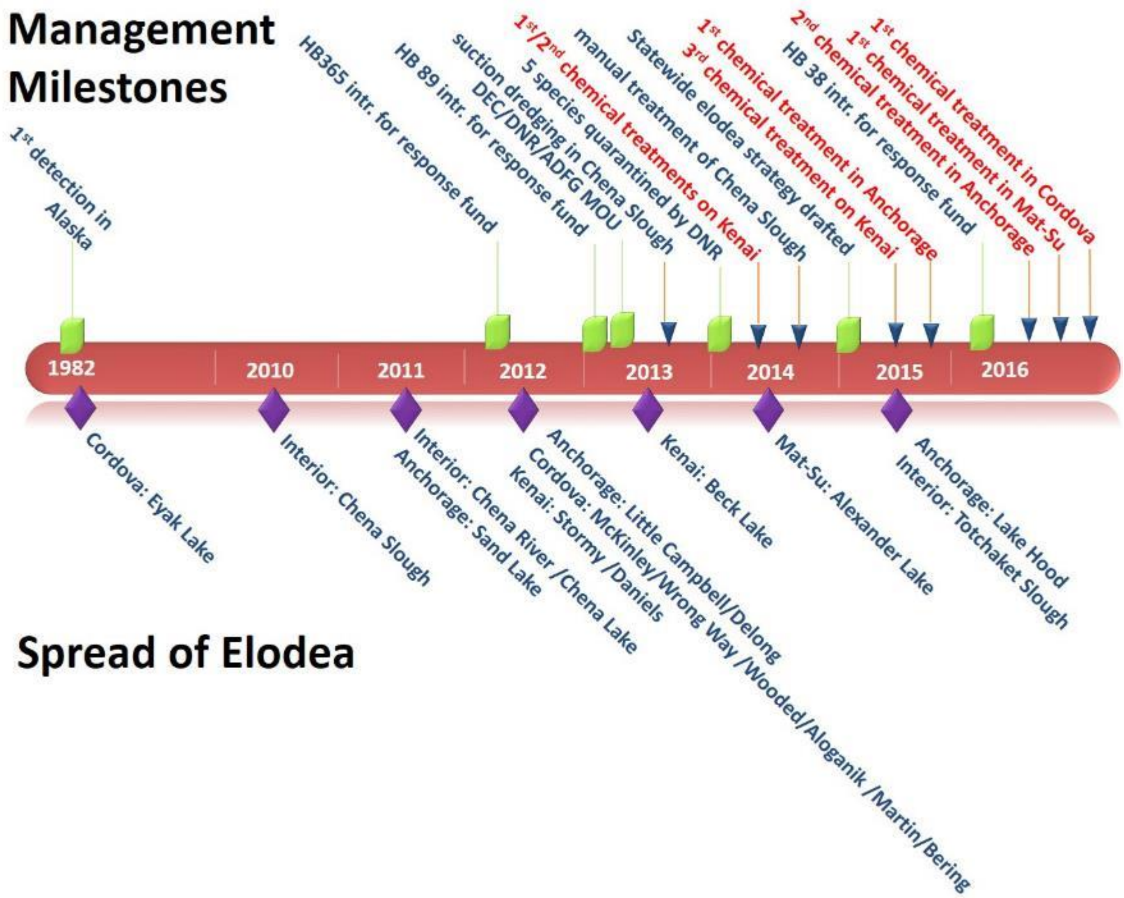

Figure 1.3 Timeline of discovery and management actions for elodea-infested waterbodies in Alaska. Source: John Morton, FWS

Among the alarming trends of long-distance dispersal of elodea across the state, 2015 also had its success stories with the completion of chemical treatment of three lakes on the Kenai Peninsula (J. Morton 2016). With budget remaining, the Kenai Peninsula Borough government decided to invest remaining funds for immediate chemical treatment of Lake Hood to reduce the risk of re-infestation of the Kenai lakes. Also, the recently created MOU turned out to be a critical piece to quick implementation of an emergency response. But the first success was also followed by further setbacks in 2016. In just two years, the infestation in Alexander Lake grew from ten acres observed in 2014 to 500 acres in 2016 (Figure 1.4).[1] The initially planned partial lake

[1] The explosive growth pattern has been documented elsewhere and underlines the need for timely and cost-effective action (Jones, Eaton, and Hardwick 1993; Leung et al. 2002).

treatment for 2016 was estimated at $96,000 in product cost, now requires a budget of $500,000 for a full lake treatment (Stewart 2016). 2016 also saw trial chemical treatment in one of the infested waterbodies in the Cordova area and eDNA sampling as well as large scale monitoring efforts being conducted by the National Park Service and U.S. Fish and Wildlife Service. After two failed attempts, a bill establishing a response fund was once again introduced to the 29th Alaska Legislature.

Figure 1.4 Rake samples (top) from elodea beds (bottom) in Alexander Lake, June 2014 (left) and June 2016 (right). Source: Heather Stewart, DNR

Different management agencies and implementing organizations are working on elodea infestations across the state. Successful eradication on the Kenai was made possible through effective leadership and public private partnership within the Kenai Peninsula Cooperative Weed Management Area.[2] Among the partners with major involvement were the U.S. Fish and Wildlife

[2] Cooperative Weed Management Areas are voluntary public private partnerships between resource management agencies, tribes, private land owners, conservation organizations, and other interested stakeholders. Their goals are to prevent the introduction and spread of invasive plants, implement effective and economically feasible management action, facilitate cooperation among managing and implementing stakeholders, and educate the public about invasive plants. There are five cooperative weed management areas in Alaska: Fairbanks, Anchorage, Kodiak Archipelago, Kenai Peninsula, and Juneau.

Service, Homer Soil and Water Conservation District[3] and Kenai Peninsula Borough government. In Anchorage, mainly DNR and the U.S. Fish and Wildlife Service have taken a leadership role and started chemical treatment of all infested waterbodies. Alexander Lake which is located on state land is chemically treated by DNR. The infestations in the Cordova area are managed by the U.S. Department of Agriculture's Forest Service, and the infestations in Fairbanks are lead by the Fairbanks Cooperative Weed Management Area and Fairbanks Soil and Water Conservation District.

1.3 Problem statement

The recent discovery of Alaska's first documented submerged freshwater aquatic invasive plant, *Elodea spp.* (elodea) motivated this study. The plant was found in Anchorage's Lake Hood, the world's busiest floatplane base where it created a pathway to spread the plant to remote freshwater landing sites (Hollander 2015b). Since Alaska has vast freshwater resources supporting the world's largest wild salmon fisheries, the spread of elodea raises concern about impacts to local salmon fisheries and freshwater resources (Carey et al. 2016). Also, the explosive and dense invasive plant growth creates safety hazards for pilots and can prevent pilots from accessing private property and recreation opportunities (CH2MHILL 2005). Given the urgency of statewide management action, quantitative information on the risk of elodea to the state's economy is critical for decision-making. Additionally, the study is inspired by the demand for more sophisticated tools informing active risk management in Alaska. The study serves as a stepping stone towards a more pro-active risk management approach for elodea and other invasive species yet to arrive in Alaska.

1.3.1 Appropriate management tools

Currently, local resource management agencies remain reliant on an invasiveness ranking system providing little information on strategic decision making (Carlson, Lapina, and Shephard 2008).In this system, experts (assessors) provide numeric scores and supporting documentation for different risk categories including qualitative ratings for establishment, ecological impact,

[3] Soil and Water Conservation Districts are government entities established under state law to provide technical assistance for the protection of land and water resources in their designated local areas. In Alaska there are twelve conservation districts.

dispersal ability, and management options (Carlson, Lapina, and Shephard 2008). After peer review, a score between 0 and 100 is calculated. Elodea's current score of 79 is in the top 10% of all listed terrestrial and aquatic plant species in Alaska (Nawrocki et al. 2011). Similar ordinal scoring systems are used elsewhere (Pheloung, Williams, and Halloy 1999; Warner et al. 2003; Hiebert and Stubbendieck 1993).

While such scoring systems inform resource managers about the relative risk for numerous species, it falls short of providing information on the absolute risk in specific on how catastrophic the invasion of a certain species can be. By ignoring the potential consequences, the ranking fails to informing decisions on whether to take or not to take action. More specifically, a single index number prevents further integration into decision analysis or damage assessments as would be achieved by a probabilistic measure and economic valuation of ecosystem services that are at risk. In addition, the discrete score stops short of informing managers about the quality of the assessment, particularly problematic with very small assessor groups. Bioeconomic risk analysis on the other hand, provides economic reasoning and information on the return on investment which can lead to actual conservation investments.

1.3.2 Probabilistic expert knowledge

Evidence-based decision making in natural resource management is frequently hampered by a lack of relevant quantitative information, particularly when timely action is necessary to avoid damage to native ecosystems and local economies. For the invasive species issue quick decisions can minimize long-term costs but concrete evidence about the invader's impacts on the ecology and economy is often limited or non-existent. In such instances, resource managers tend to address risk qualitatively or even haphazardly mixing facts with values instead of being able to follow a data-driven approach that separates evidence from the perceived values at stake (Maguire 2004). In the presence of significant uncertainty, formally quantifying expert opinion can give structure to a more substantiated decision process and is particularly useful when a broader knowledge base is needed, for example in predicting extreme events in invasion ecology (Franklin et al. 2008; Willis et al. 2004).

Recent improvements to probabilistic elicitation in the natural sciences focus on complementing probabilistic expert knowledge with artificial intelligence to detect and correct bias (Sikder, Mal-Sarkar, and Mal 2006; Regan et al. 2005). While these improvements have enhanced elicitation, they have not addressed a more structured need to quantify expert opinion and minimize bias “by design.” In the social sciences, the validity of direct probabilistic reasoning has long been debated (Morgan and Henrion 1990; Tversky and Kahneman 1974). Opponents believe that knowledge about a subject area doesn’t readily translate to an ability to accurately convey knowledge in probabilistic terms, particularly for highly uncertain events. Experts often express their knowledge in words rather than numbers and their attempts to assign numerical values result in heuristics and biases (Saaty 1990; Tversky and Kahneman 1974).[4] Such limitations can lead to more costly and lengthy elicitation procedures and extensive training. As a result, the expert pool remains limited by experts familiar with probability encoding or experts who have the time and willingness to receive training. Small expert samples are more likely to create doubt in reliable results for interpretation and decision making (Yamada et al. 2003).

The application of a discrete choice methods (DCM) for expert elicitation avoids many of these issues and provides additional ways to analyse and apply expert opinion beyond what traditional methods can achieve. Respondents simply choose among discrete alternatives (environmental scenarios) within a binary question format that asks for an existential statement about a believed outcome (e.g., state of nature) based on a set of attributes (e.g., habitat characteristics). Attributes vary across two or more alternatives and together form a choice set. In this way, experts focus on the ecological relationships between attributes. Doing so avoids experts having to translate their knowledge into probabilistic terms, yet probabilities estimated indirectly from the choice data are ultimately available for decision analysis.

1.4 Research objectives

The overarching research objective of this study is to inform resource managers about the potential range of future economic damages to fisheries and recreationists at risk from elodea invasions and to provide strategic guidance on when and where to intervene. The second

[4] In real life, humans subconsciously weigh attributes in their decisions. As the number of rank order tasks increases, respondents apply simplification and elimination strategies that lead to bias and validity concerns (Louviere 1988). Similar issues arise with ratings data that require strong assumptions on the order of preferences to measure them (Louviere 1988). Despite improvements to the ranking exercise through for example the Analytic Hierarchy Process (AHP) or Maximum Difference Scaling (MaxDiff) serious theoretical issues remain such as rank reversal and limitations on the number of rank items (Ishizaka and Labib 2009)

objective is to develop a comprehensive bioeconomic risk analysis tool that can be used for risk management of new arrivals and evaluate investments into prevention. The following specific research objectives are guiding the following chapters:

1. Account for uncertainty in elodea's potential effects on salmon persistence and productivity in Alaska by applying different methods to elicit, quantify, and analyse probabilistic expert opinion.
2. Estimate potential region-specific market and non-market economic damages to multiple stakeholders across Alaska.
3. Understand and model the floatplane pathway of distributing elodea from urban source locations to remote waterbodies across Alaska.
4. Provide recommendations for optimal management based on cost benefit calculations related to the set of management actions.

This report is outlined as follows. Chapter 2 borrows techniques from marketing research to quantify expert's knowledge about elodea's potential effects on Alaska's salmon resources. Chapter 3, uses structured expert judgment and market valuation techniques to assess the potential damages of elodea to commercial sockeye salmon fisheries. For Chapter 4, a survey with floatplane pilots residing and operating in Alaska was conducted. The survey asked about home base and destination locations as well as frequency of flights and information about costs of floatplane operation. An econometric model of pilot behavior was developed to estimate non-market values that are at stake should elodea invade floatplane destinations. Chapter 5 integrates the previous chapters for a comprehensive bioeconomic risk analysis. In specific, this chapter evaluates the net benefits associated with a set of possible management options and provides recommendations for optimal management of elodea cross Alaska.

2 Quantifying expert knowledge about the persistence of salmonids in elodea-invaded habitat: Applying a discrete choice experiment for expert elicitation

This study elicits expert belief on the persistence of salmon populations in elodea-invaded salmon habitat to improve understanding of the potential risk posed by elodea for five local species of salmonids: sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), chinook (*O. tshawytscha*), dolly varden (*Salvelinus mulmu*), and humpback whitefish (*Coregonus pidschian*). A discrete choice model (DCM) is used to analyse expert knowledge. The approach is aimed at avoiding many of the described problems in ecological expert elicitation by broadening the expert pool and formalizing elicitation design, process, and analysis. The objectives of this chapter are to estimate probabilities indirectly for elodea invasions of salmon habitat. The DCM determines whether expert evidence suggests that different fish species respond differently to elodea invasions and what the underlying habitat factors may be. Results indicate that most experts believe elodea will have negative effects on salmonids with varying levels of certainty. The analysis quantifies the relative importance of habitat attributes to salmon fitness and the estimated probability of salmon persistence following elodea invasion.

2.1 Methods

2.1.1 Discrete choice model of expert opinion

DCMs have often been applied to understand public preferences, values, decision making, and trade-offs (Hoyos 2010; Louviere et al. 2007; Knowler et al. 2009). Less frequently, DCMs have been used to analyze how resource managers make decisions about wildfires, which similar to invasive species, can have catastrophic consequences if managed poorly (Wibbenmeyer et al. 2013). Even though DCM is now widely used in many different fields, it has not been effectively used for expert elicitation in ecological research. DCM assumes humans are rational decision makers. Yet, natural resource managers tend to be risk averse over low probability losses and high probability gains, and risk seeking over high probability losses and low probability gains (Wibbenmeyer et al. 2013). This irrational behavior can lead to resource misallocation and economic loss (Shaw and Woodward 2008).

The goal of DCM is to measure influence of attributes on choices among alternatives accounting for as much heterogeneity as possible between individuals and groups of experts (McFadden 1973; Ben-Akiva and Lerman 1985). To achieve that goal, the random utility model (RUM) defines overall utility of an alternative j to individual n as U_{nj}, comprised of observable utility V_{nj} and the unobservable utility, ε_{nj} thus $U_{nj} = V_{nj} + \varepsilon_{nj}$ (McFadden 1973). The measured component of utility in linear form for individual n is $V_j = \beta_{0j} + \beta_{1j} f(X_{1j}) + \beta_{2j} f(X_{2j}) + ... + \beta_{kj} f(X_{kj})$, where β_0 represents the average of all the unobserved sources of utility, $\beta_{1,...,k}$ is the coefficient or part-worth that estimates the contribution of attribute $X_{1,...,k}$ to the observed sources of relative utility (Hensher, Rose, and Greene 2005).[5] X_1 is the first attribute in k number of attributes.

The choice rule states that each individual evaluates all alternatives presented to him/her, U_j for $j = 1,..., J$ alternatives in the choice set, then compares $U_1, U_2, ..., U_J$ and finally chooses the alternative with maximum utility $\max(U_j)$. The probability of an individual respondent choosing alternative i is equal to the probability that the utility associated with alternative i is equal or greater than the utility of any other alternative, U_j , in the choice set, thus $p_i = p(U_i \geq U_j)$, where $i \neq j$ and $j \in j = 1,..., J$. Since utility is comprised of an observable and unobservable part, the choice rule becomes a random utility maximization rule. The probability of i being chosen is equal to the probability that the difference in unobservable sources of utility is less than or equal to the difference in observable utility for alternative i after the respondent evaluates all the other alternatives, thus $p_i = p(\varepsilon_j - \varepsilon_i) \leq (V_i - V_j)$ where $i \neq j$ and $j \in j = 1,..., J$.[6] Within the choice model, the unobserved components of utility related to an individual selecting

[5] Note, the utility level measured is “relative” to the utility levels associated with all other alternatives shown in the choice set. Thus, there exists a base reference utility within the choice set but not across choice sets, preventing comparison of absolute utility for an alternative calculated in one choice set with another choice set (Hensher, Rose, and Greene 2005). The base reference utility is also called “scale of utility.”

[6] One could incorporate information on the second, third, etc. most preferred alternative, recognizing that there is useful information in having respondents rate the alternatives. This analysis will focus only on one preferred choice.

an alternative is treated as a random piece of information, uncorrelated with all other alternatives but having the same variance across alternatives (Louviere, Hensher, and Swait 2000). The probability that expert n chooses alternative, i, from a set of J alternatives presented in a choice set equals the multinomial logit (MNL) specification:

$$p_{ni} = \frac{e^{\beta_n X_{ni}}}{\sum_{j=1}^{J} e^{\beta_n X_{nj}}} \ , \tag{1.1}$$

where $i \neq j$ and $j \in j = 1,...,J$ (McFadden 1973).[7]

2.1.2 Hierarchical Bayesian Estimation

A hierarchical Bayesian (HB) approach is used to estimate individual-level coefficients by drawing from a multivariate normal distribution described by a vector of means, α, and a matrix of covariances, D. The means of the individual part-worths, α, are assumed to be normally distributed with a mean equal to the average part-worths and covariance matrix equal to $\frac{D}{N}$, with N being the sample size (Orme 2009b). Estimates of D are drawn from an inverse Wishart distribution.[8] Monte Carlo integration applying the Metropolis Hastings algorithm and Gibbs sampling draw conditionally from the joint posterior distribution and simultaneously estimate the parameters α, β, and D. With these estimates in hand, individual utility distributions are derived (Gelman et al. 2013; Orme 2009b).[9] The part-worths are a compromise between the aggregate

[7] In the MNL, the random component of utility, is assumed to be IID, independent (uncorrelated alternatives) and identically distributed (across all j the distribution explaining has constant variance). It assumes that the ratio of probabilities of any two alternatives cannot change if any other alternative is added or taken away from the set of alternatives in a choice set, meaning all pairs of alternatives are equally similar or different. However, if there is sufficient data quality that minimizes the amount of unobserved heterogeneity, the IIA assumption has small consequences (Hensher, Rose, and Greene 2005).

[8] Note, the Wishart distribution is the conjugate prior of the covariance matrix of the multivariate normal distribution and is commonly used to deal with large dimensionality.

[9] The draws from the joint posterior distribution after convergence, quantify uncertainty in the each respondent's utility estimate. The same can be shown using the historic draws for α for the entire sample.

distribution of beliefs across the sample and the individual's belief, and result in a conditional estimate of the respondent's parameters. The posterior distribution of individual parameter estimates allows for an assessment of uncertainty in expert belief.

Through Equation 2.1, the individual expert's choice probabilities for given alternatives describing invaded salmon habitat are calculated. The sum of individual choice probabilities for a given alternative are equal to the proportion of times that alternative was chosen over all other alternatives. This proportion can be calculated for individual experts, groups of experts, or the entire sample and can be interpreted as the subjective probability related to the state of nature described by the alternative.[10]

Simulation allows for a detailed look at the sensitivities of choice probabilities in relation to the attribute levels that form alternative habitat hypothesis. This analysis reveals the relative importance of attribute levels, interactions between attributes, and the degree of disagreement between individuals and groups of experts.[11] Two choice simulation approaches are most commonly used, 'randomized first choice' and 'share of preference.' The latter assumes respondents carefully evaluate each alternative which is most appropriate for this analysis and the former assumes less observant choice behavior (Train 2003).[12]

2.1.3 Study design

The iterative design process started with an extensive literature review and key informant interviews to refine the problem, identify attributes and levels, establish alternatives, and finally

[10] The economics literature refers to this proportion commonly as market share (Hensher, Rose, and Greene 2005; Train 2003).

[11] The scale factor can be used in simulation to adjust the choice shares to the true shares if they are known (Hensher, Rose, and Greene 2005). The exponent of the scale factor by default is set equal to 1 and usually is adjusted downward (B. Orme 2009a).

[12] It assumes that the utility maximizing alternative, i, is equal to $U_i = X_i(\beta + \varepsilon_A) + \varepsilon_p$, where ε_A adds attribute variability accounting for similarity relationships and ε_p adds alternative variability which dims the latter effect. The probability of choosing alternative i in choice set S is equal to the probability that the randomized utility draw is largest compared to the utility draws for all the other alternatives, or mathematically $p(i \mid S) = p(\mathrm{U}_i \geq U_j)$ for all j in S. The simulation draws random U_i 's and sums the probabilities for a specified alternative, i by individual expert, a group of experts, or for the entire sample.

consider and generate the design (Hensher, Rose, and Greene 2005). Discrete choice experiment software facilitated the final experimental design and data collection (Sawtooth 2016b). Particular design criteria included maximum variation in attribute levels within choice sets (minimal overlap), equal representation of attribute levels (level balance), and approximately equal choice probability of alternatives (utility balance) (Rich Johnson, Huber, and Bacon 2003).

In this study, experts' "choice" task was to select the alternative salmon habitat s/he believed results in long-term persistence of salmon populations compared to other alternative habitat presented in each choice set. Persistence is defined as the presence of a functionally viable local salmonid population for at least 20 years (Peterson et al. 2008). Attribute levels cover extreme values (end-points) that are potentially outside the range with which experts are familiar. In this way, the design is more likely to cover the actual values of changing environmental attributes given a perturbation through invasive species.[13]

Attributes and attribute levels were selected based on the literature review aimed at a broad overview of ecological effects that key informants vetted to be related to the viability of salmonid populations in invaded habitat. Given the relative lack of research examining the effects of aquatic invasive species on salmonid habitat, both local and non-local sources of literature were used. The mean vegetation cover observed in Alaska is around 27% in uninvaded lakes and reaching 100% in invaded waterbodies (Rinella et al. 2008; Lane 2014). Elodea can increase dissolved oxygen (DO) in the upper parts of the plants to 9 mg/l but DO concentrations within 5 cm of the substrate can reach as low as 0.4 mg/l (Spicer and Catling 1988). Additionally, frequent die-back events can lead to perturbation of the entire lake ecosystem with very low DO concentrations during die-back (Barko and James 1998; Jeppesen et al. 1998; Burks, Jeppesen, and Lodge 2001; Sebastian Diehl et al. 1998).

Invasive aquatic plants can also indirectly affect fishes through food web effects with complex and uncertain outcomes (Erhard, Pohnert, and Gross 2007; Schultz and Dibble 2012). Mean macroinvertebrate abundance counts in Alaska lakes range between 374/m^2 and 1125/m^2. Zooplankton biomass in Alaska sockeye nursery lakes ranges between 22 mg/m^2 and 2223 mg/m^2 (Edmundson and Mazumder 2001). The wide range in food availability related to invaded

[13] The linearity assumption between the end-points limits valid extrapolation to within the range of attribute levels. The benefit of a simpler and more efficient design however outweighs this limitation.

habitat were chosen to reflect the uncertain effects of this attribute for salmonids. Lastly, elodea beds provide habitat for predatory northern pike (*Esox lucius*) and have the potential to cause synergistic interactions leading to accelerated impacts on native ecosystems (Casselman and Lewis 1996; Simberloff and Holle 1999). Invasive pike in Southcentral Alaska can reach densities of up to 36 pike per surface acre (Sepulveda et al. 2014; Sepulveda et al. 2013).

Two critical design criteria are important to the invasive species case. First alternative-specific attribute levels reflect the ecological distinction between the invaded and un-invaded state of habitat. Second, unambiguous *a-priori* preference order in the attribute levels constrain the order and sign of estimated coefficients to be consistent with ecological expectations. For example, by design, more DO, more prey, and less predation is better for salmon. This approach is also known as cardinal utility, a framework applied to decision making under uncertainty and allows the assumption of rational choice to be upheld (Table 1.1) (von Neumann and Morgenstern 1947).[14]

By design, ambiguity and other linguistic uncertainty are minimized to maximize interpretation, usefulness, and accuracy of the expert elicitation. To achieve this, a background document clearly defines each attribute included on every page of the questionnaire (Supplemental file).[15] While reducing ambiguity means decreasing unobserved heterogeneity in the observed choice behavior, the careful selection of attributes also minimizes hypothetical bias arising from lack of context, when the hypothetical situation differs from real world situations (von Gaudecker, van Soest, and Wengström 2012).

[14] Hierarchical Bayes estimation applies constraints on orders of part-worths within attributes, and enforces signs for linear coefficients. The constraints are applied for all respondents. They are useful if the goal of the estimation is individual level coefficients; however, this technique is less applicable to situations where researchers are interested in predicting choice shares for the population (B. Orme 2009a). An estimation process called ‘simultaneous tying’ is geared towards achieving both of these goals (R. Johnson, Huber, and Bacon 2003). The current results presented below estimate utilities without constraints as these can reduce variance at the expense of increasing bias.

[15] The background document further described the elicitation task and was accessible to respondents on all pages of the online elicitation questionnaire.

Table 2.1 Attributes and levels

Attribute	Uninvaded habitat		Invaded habitat	
	Level 1	Level 2	Level 1	Level 2
Vegetation type and cover (%)[c]	Indigenous 0%	Indigenous 50%	Elodea 50%	Elodea 100%
Dissolved oxygen (mg/l) [a, c]	5.5	10.5	0.5	5.5
Prey abundance (mg/m^2) [a, c, d]	400	600	30	3000
Piscivorous fish (#/acre) [a, c]	5	20	20	35
Location of aquatic vegetation [b]	backwater, lake, entire habitat range			
Salmon species [b]	sockeye, coho, chinook, dolly varden, humpback whitefish			

a) Attributes that have unambiguous a-priori preference order. b) Non-alternative specific attributes.
c) Alternative-specific attributes dependent on the State of habitat variable (uninvaded or invaded). d) For sockeye mg/m^2 zoopkankton, all other macroinvertebrates $/m^2$.

Identifying which alternatives should be included in the choice sets is one of the design challenges[16] that was overcome by deploying an adaptive choice design where respondents design alternatives themselves ("adaptive choice-based conjoint" ACBC) (Rich Johnson, Huber, and Bacon 2003; Orme 2009a). This customized design results in smaller required sample sizes and better estimator performance (Cunningham, Deal, and Chen 2010). Before the customized choice sets are presented (Figure 2.1), the respondent is asked to complete design tasks that inform individual-level prior information for constructing an efficient design "on the fly" (Table 2.1).[17,18] ACBC designs result in choice sets that are more relevant to respondents resulting in better choice data. ACBC is particularly useful for predictive purposes under small sample sizes and as such applicable to expert elicitation (Low-Choy, O'Leary, and Mengersen 2009). The

[16] In the worst case, the presented choice sets are not part of the respondent's beliefs or preferences resulting in additional unexplained utility.

[17] The BYO task asks experts to select habitat characteristics they believe most likely support the persistence of salmonids. In a subsequent screener section, attribute combinations are clustered around the BYO and respondents are asked to select which habitat alternatives are a possibility for salmonid persistence. Further probing questions identify attribute levels that are either 'unacceptable' or a 'must have' and inform the design about respondent-specific cut-off rules. Alternatives selected as possibilities in the screener section are carried forward into the final choice sets used for estimation.

[18] When estimating part-worth utilities collected through an adaptive choice model, the assumption of generic HB is that the three sections do not vary in scale even though in reality the BYO has larger scale than the other two. The "Otter's Method" is used to account for the difference in scale during HB estimation (Howell 2007).

design is simulated using robotic respondents resulting in D-efficiency of 75%. Table 2.2 shows the final structure of the questionnaire.[19,20]

Table 2.2 Structure of discrete choice questionnaire

1. Introduction and consent
2. Background reading material
3. Question about expertise
4. Build your own scenario
5. Screener tasks
6. Unacceptable task
7. Screener task
8. Unacceptable task
9. Must have task
10. Screener
11. Unacceptable
12. Must have
13. Screener
14. Unacceptable
15. Must have
16. Screener
17. Unacceptable
18. Must have
19. Screener
20. Final choice tournament
21. Rating of overall effect
22. Open ended comment

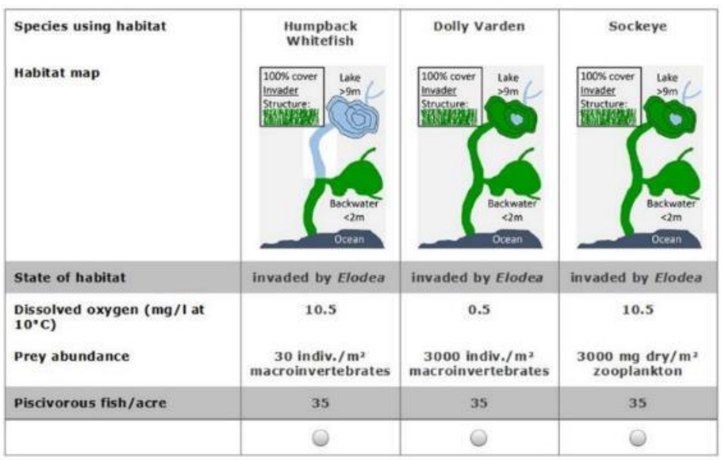

Figure 2.1 Example of a choice set presented to an expert in the final choice tournament containing ten choice sets.

The purpose of the design tasks in ACBC is to minimize bias and heuristics by design. To mention a few, social desirability bias is minimized by the complexity of the choice task placing

[19] The number of screener tasks, unacceptable, and must haves was determined following software suggestions (Sawtooth 2016b).

[20] Subsequent application of the choice experiment approach to expert elicitation could use hold out tasks to improve and validate expert opinion. A hold out task is a fixed choice set that is given to each respondent but is not part of the estimated model, instead is used to test the estimated model against choices in the hold out task (Howell 2007).

additional burden by showing a particular attitude through conscious selection of choices (Ding and Huber 2009). Availability bias is reduced through the various alternative habitat scenarios that experts are confronted with, inhibiting them from easily retrieving judgments from memory. Representativeness is diminished because the choice task requires experts to think about functional relationships between attributes rather than about similarities. The issue of anchoring is irrelevant because experts are not required to translate their knowledge into probabilistic terms (Tversky and Kahneman 1974). Finally, the screener tasks act like probing questions that keep experts accountable for their choices, which reduces overconfidence. Finally, particular attention is given to the visual and tabular format of the discrete choice task to minimize filtering heuristics (Hoehn, Lupi, and Kaplowitz 2010). The questionnaire presents alternatives through hypothetical habitat maps, specifying stream depth and gradient to further limit ambiguity.

A pre-test with 20 arbitrarily selected experts resulted in 12 responses and led to various rounds of revisions to eliminate ambiguities. Several experts suggested eliminating calibration tasks included in an earlier version. These respondents felt the calibration exercises were upsetting and questioned their credibility. The change reduced respondent burden to about 45 minutes. Final data were collected in March and April of 2015.

The expert pool was identified using an extensive literature review of 296 sources and was comprised of individuals with substantive knowledge of Pacific salmonids in freshwater habitat, the ecological role of submerged aquatic vegetation, or invasive freshwater aquatic plants. Expert selection followed common guidelines for expert elicitation (Martin et al. 2012; Drescher et al. 2013; ACERA 2010) and were selected based on the number of citations in peer-reviewed publications using Google Scholar. Due to the localized issue of elodea in Alaska, the expert pool was expanded to include state and federal resource management agencies with job titles including fishery biologist, fisheries scientist, fish habitat biologist, and invasive species specialist. These individuals brought knowledge on localized variability and local observations to the expert pool of 111 contacts.[21]

[21] Research on identifying experts for ecological and resource management issues is an ongoing field of study with no clear guidance and available definitions (Drescher et al. 2013; Martin et al. 2012).

2.2 Elicitation results

2.2.1 Base case

A total of 56 experts responded for a response rate of 50%.[22] The sample is representative of the initial expert pool (Table 2.3). Alaska residents and experts with both salmon and other invasive species expertise were more likely to respond. Concentrated local knowledge and oversampling of salmon expertise can be viewed as desirable rather than a source of selection bias (Drescher et al. 2013). The inclusion of non-local experts aims at minimizing motivational bias that occurs when experts have personal stakes in the ecological issue (Martin et al. 2012; Drescher et al. 2013).

Table 2.3 Sample representativeness

Expertise	Initial expert pool		Respondents	
Salmon	82	74%	45	80%
Aquatic vegetation	38	34%	18	32%
Salmon and other fishes	9	8%	7	13%
Alaska-based	80	72%	46	82%
Total	111		56	

Along with results for the entire sample the MNL model was estimated for groups of experts separately. This approach allows for a more detailed look at choice consistency and how belief within and among experts varies. Results present expert belief using three different measures: part-worth utilities, individual expert choice probabilities, and choice probabilities related to groups or the entire sample. Part-worth utilities for each individual expert were estimated using HB with a burn-in of 10000 iterations before 1000 random draws were saved. These are then used in the Sawtooth Choice Simulator to derive choice probabilities (Sawtooth 2016a).

Table 2.4 presents the estimated MNL model and mean part-worth utilities for explanatory variables affecting the choice response variable: believed persistence of salmonids. To ensure that alternative specifications are not affecting part-worth utilities, the results have been rescaled

[22] Despite the very good response rate there is a possibility that the responses are not representative of all the available expertise. A non-respondents questionnaire was not conducted, preventing non-response bias to be explored.

to zero-centered utility differences. Thus, attributes can be compared so that the average value for utilities in each attribute is zero and the total sum of utility differences between the best and worst attribute level across attributes is equal to 100 times the number of attributes. The estimated mean part-worths change with the number of respondents and weights in the simulation. Signs show directional effects such that positive signs indicate a positive contribution to salmonid persistence and the mean part-worth value indicates the magnitude of the effect. The standard deviation and 95% confidence intervals provide a measure of disagreement across the expert sample and serve as uncertainty indicators. The model is twice as good as a model of chance in predicting expert choices shown by the root likelihood (RLH) of 0.717 (Table 2.4).[23]

Table 2.4 Part-worth utility distributions for believed salmon persistence

Attribute level		Mean	SD	95% CI	
State of habitat					
	Elodea-invaded	-129.95	40.49	-140.56	-119.35
	Indigenous	129.95	40.49	119.35	140.56
Species					
	Sockeye	8.85	21.99	3.09	14.61
	Coho	10.89	32.01	2.50	19.27
	Chinook	-12.28	36.52	-21.85	-2.72
	Dolly Varden	1.42	28.94	-6.16	9.00
	Whitefish	-8.88	30.73	-16.93	-0.83
Location of vegetation					
	Backwater	16.07	31.40	7.85	24.30
	Entire system	-14.20	22.94	-20.21	-8.19
	Lake	-1.87	24.97	-8.41	4.67
Vegetation cover [a]					
	50% invaded	39.67	35.90	30.27	49.07
	100% invaded	-39.67	35.90	-49.07	-30.27
	0% indigenous	0.35	38.44	-9.72	10.42
	50% indigenous	-0.35	38.44	-10.42	9.72
Dissolved oxygen (mg/l) [a]					
	0.5 invaded	-98.90	74.80	-118.49	-79.31
	10.5 invaded	98.90	74.80	79.31	118.49
	5.5 indigenous	-52.67	53.77	-66.75	-38.58

[23] The RLH of 0.717 is compared to the the RLH of the three alternatives shown in the final choice tournament which equals 0.33, the geometric mean of the predicted probabilities. Other indicators for goodness of fit include the average variance of part-worths and the root mean square (RMS) of all part-worths for the sample.

10.5 indigenous	52.67	53.77	38.58	66.75
Prey abundance (mg/m^2) [a]				
30 invaded	-35.05	29.08	-42.67	-27.44
3000 invaded	35.05	29.08	27.44	42.67
400 indigenous	-10.59	19.01	-15.57	-5.61
600 indigenous	10.59	19.01	5.61	15.57
Piscivorous fish (#/acre) [a]				
20 invaded	15.98	20.45	10.63	21.34
35 invaded	-15.98	20.45	-21.34	-10.63
5 indigenous	48.06	46.39	35.91	60.21
20 indigenous	-48.06	46.39	-60.21	-35.91
No. of observations	560			
No. of respondents	56			
No. of parameters	26			
Pseudo R^2	0.576			
Root Likelihood (RLH)	0.717			
Average Variance	1.387			
Parameter root mean square	1.630			

a) Alternative-specific attribute levels dependent on state of habitat.

The state of habitat attribute is the most important as it specifies expert belief in persistent salmonid populations in either invaded or uninvaded habitat. There is wide agreement among experts that uninvaded habitat results in persistent salmon populations and invaded habitat threatens that persistence (Table 2.4). Through simulation, the probability an expert selects an invaded habitat believed to result in persistence can be analyzed using the base-case assumptions outlined in Table 2.5. Figure 2.2 illustrates the distribution of individual choice probabilities related to invaded habitat with a median choice probability of 0.04 indicated by the dashed line and a mean of 0.21 (dotted line). In other words, half of the expert sample believes that the probability that salmon can persist in elodea-invaded habitat is less than 0.04, whereas the average expert believes that the probability is 0.21.

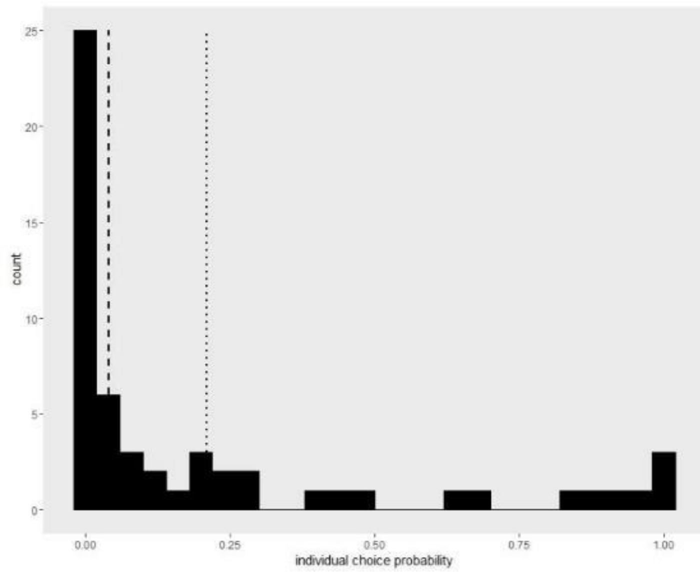

Figure 2.2 Histogram of individual choice probabilities of salmonid persistence in invaded habitat. Sample median (dashed) and mean (dotted line).

For the species attribute, the relatively low magnitudes for mean part-worths show that experts generally think that all species are equally vulnerable to elodea (Table 2.4). However, a closer look at how the estimated choice probabilities for persistence vary among experts reveals insights on experts' outlook for specific salmonid species in general. Figure 2.3 illustrates that experts were proportionally less likely to select alternative habitat occupied by Chinook salmon (*Oncorhynchus tshawytscha*), humpback whitefish (*Coregonus pidschian*), and Dolly Varden (*Salvelinus malma*) compared to sockeye (*Oncorhynchus nerka*) and particularly coho (*Oncorhynchus kisutch*).

Table 2.5 Base-case habitat alternative

	Invaded		Indigenous
Attribute	Base level	Sensitivity range	Base level
Vegetation cover (proportion)	0.5	0.5, 1.0	0.5
Dissolved oxygen (mg/l)	5.5	0.5, 10.5	5.5
Prey abundance (mg/m^2) [a]	400	30, 3000	400
Piscivorous fish (#/acre)	20	20, 35	20

a) For salmonids other than sockeye, the unit is abundance of prey/m^2.

Similarly, the *location of aquatic vegetation* attribute does not receive much weight in experts' choices among alternative habitats. The presence of aquatic vegetation in the backwater location, regardless of invasion status, is considered a positive contribution to persistence, whereas alternatives showing aquatic vegetation in all parts of salmon habitat or in the lake are believed to be negative. Experts agree about the negative effects of aquatic vegetation in the lake location compared to the positive effects of the backwater location (Figure 2.4).

The extent of vegetation cover is particularly important to experts in invaded habitat and much less so under uninvaded conditions. However, experts disagree more on the effect of vegetation in uninvaded habitat than the extent of vegetation in invaded habitat (Table 2.4). For DO and prey abundance, the directional effects are as expected, with higher DO and prey levels showing positive effects on persistence (Table 2.4).

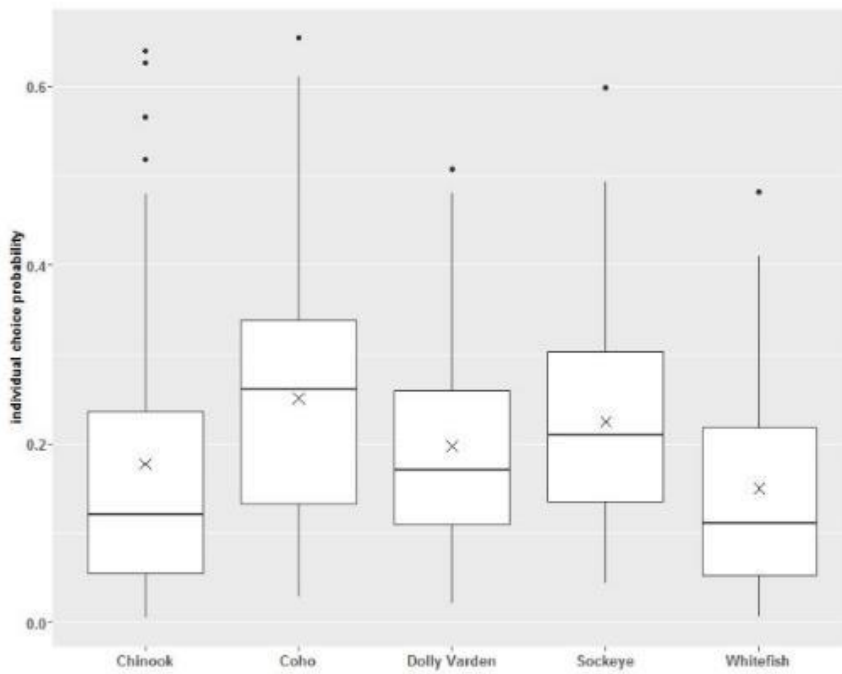

Figure 2.3 Boxplot comparing persistence probabilities across species regardless of elodea presence. Lower and upper quartile (box), median (bold line), mean (x). A probability of 0.2 represents equal expert choice across species.

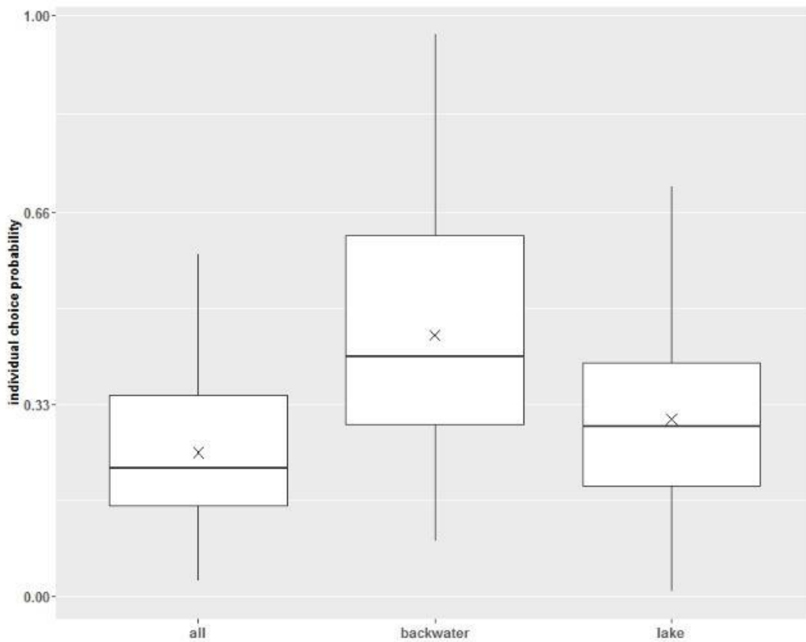

Figure 2.4 Boxplot of individual choice probabilities related to the effect of vegetation location on salmonid persistence regardless of elodea presence. Lower and upper quartile (box), median (bold line), mean (x). A probability of 0.33 represents equal expert choice across locations.

2.2.2 Sensitivity analysis

A simulation of invaded habitat under varying environmental conditions shows the sensitivity of choice probabilities across the expert sample (Figure 2.5). For each alternative-specific attribute related to invaded habitat, choice probabilities are estimated across a sensitivity range described in Table 2.5. DO levels have the largest marginal effect on choice probability, indicating that at 0.5 mg/l, the mean choice probabilities of salmonid persistence in invaded habitat can reach below 0.1 and at 10.5 mg/l can reach up to 0.5 (Figure 2.5). The steepness of the DO curve illustrates that any directional change in DO can have large consequences on the believed persistence of salmon in invaded habitat, more so than any other attribute that was part of the design. In contrast to DO stand the marginally smaller effects of prey, elodea cover, and predation. Besides the positive effect of increasing prey abundance, increasing elodea cover and increasing predation have negative consequences for salmon with elodea cover being slightly larger in magnitude.

The uncertainty among experts around the estimated mean choice probabilities vary slightly within and across attributes as illustrated by the varying width of the confidence intervals (Figure 2.5). First, expert agreement is higher for lower DO (SE = 0.03) compared to higher DO (SE = 0.05), lower prey abundance (SE = 0.04) versus higher prey abundance (SE = 0.05), less elodea cover (SE = 0.043) versus high elodea cover (SE = 0.038). Predation levels seem to have similar uncertainty across the entire observed range (SE = 0.043, 0.04). Second, expert agreement varies across attributes showing more disagreement on the effects of DO levels and extent of elodea cover compared to prey abundance and predation.

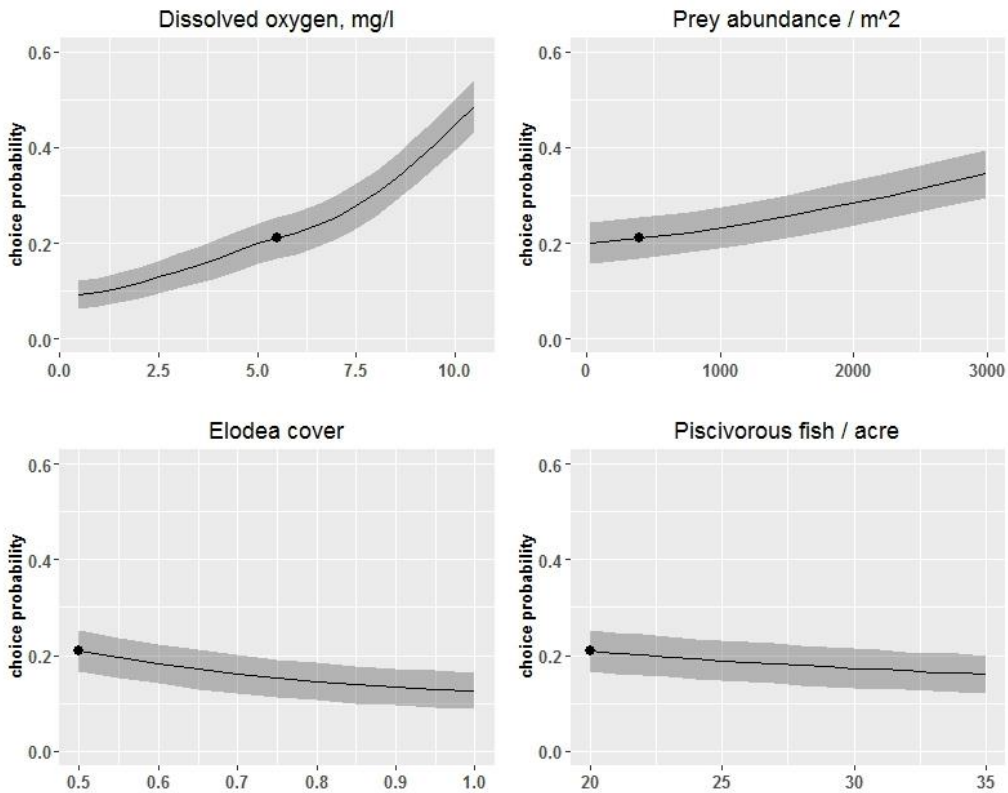

Figure 2.5 Sensitivity of choice probability of salmonid persistence in elodea-invaded habitat given changes in habitat-specific attribute levels. Sample mean (black line), 95% CI (shade), base case (dot).

The sample was divided into five expert groups to further explore the level of agreement among experts. Groups were formed according to the results of a rating exercise placed at the end of the final choice tournament (Table 2.6). Experts were asked to rate the overall effect of elodea on salmon persistence using a five point semantic differential scale ranging from significantly negative to significantly positive effect (Table 2.6).

Table 2.7 shows the mean part-worths when applying the MNL model to each group. Regardless of elodea present in a salmon system or not, experts across groups believe sockeye to be more persistent than chinook, consistent with studies that suggest large scale shifts in ocean conditions favor sockeye and other salmon species, but the outlook for chinook seems poor (Adkison and Finney 2003; Hare, Mantua, and Francis 1999). All expert groups, except group 3, believe that coho have a generally better outlook than whitefish. Group 3 favors whitefish over coho for a positive long-term outlook. The effect on persistence related to where aquatic vegetation is located within a salmon system, regardless of an invasion, is consistent with previously shown results.

Table 2.6 Experts' rating of elodea's overall effect on salmonid persistence (n=56)

Group	Overall effect on salmonids	Frequency
1	Significantly negative	10 (18%)
2	Moderately negative	35 (62%)
3	No effect	3 (5%)
4	Moderately positive	1 (2%)
5	Don't know	7 (13%)

Note, none rated elodea to have significantly positive effects.

Most importantly, all groups show a preference for uninvaded habitat, although the magnitude of the effect differs. For experts whose ratings indicate that they do not belief elodea has an effect on salmonid persistence (Group 3 in Table 2.6), habitat status (invaded/uninvaded) is given the least weight among the attributes. This result validates the DCM. However, experts in group 3 heavily weigh the extent of invasive elodea cover compared to other groups, indicating that group 3 experts consider habitat status and elodea vegetation cover among alternatives. All groups agree about the directional effect of the extent of elodea vegetation cover, where 100% coverage with invasive elodea is viewed as negative for salmonid persistence. Slight disagreements occur between vegetation cover in uninvaded habitat, where groups 1 and 5 have stronger preferences for 0% cover whereas the remaining groups believe 50% cover to be more beneficial habitat for persistent salmonids (Erhard, Pohnert, and Gross 2007; Schultz and Dibble 2012). This result may indicate that experts considered different life stages or salmon species when considering this attribute as supported by experts' final comments. In addition, varying levels of knowledge regarding the complex role aquatic vegetation plays for fishes could also play a role.

The highest level of agreement among groups occurred for the DO variable for both invaded and uninvaded habitat, with group 3 weighing DO most heavily. Additional agreement among groups are the observed directional effects of the prey abundance and predation variables in invaded habitat with one expert (group 4) heavily weighing this attribute more so than almost any other attribute except DO. This expert believes that high prey abundance in elodea beds is a strong driver of persistence, and most likely the reason why the expert rated elodea as having a moderately positive effect on salmonids. The prey and predator abundance attributes in uninvaded habitat showed various disagreements between group 1 and 2. Group 3 had opposing preferences compared to all other groups on the level of preferred predation, showing preference for higher predation levels.

Table 2.8 provides a comparison of attribute importance within and across groups. Relative importance scores are calculated as $\text{score}_k = \frac{\max \beta_k - \min \beta_k}{\sum_{k=1}^{k} \left(\max \beta_k - \min \beta_k \right)} 100\%$, where $\max \beta_k - \min \beta_k$ is the range of part-worth utilities observed across all levels of attribute k.[24] In other words they are calculated based on HB estimated utilities, averaged across the sample, and standardized to sum to 100. If an attribute has twice the score of another attribute, it is twice as important in explaining expert belief (Orme 2010). For experts in Group 1, state of habitat is the main driver of persistence and three times as important as DO and predation in uninvaded systems. Group 2 experts focused their attention like group 1 on the state of habitat variable but with more weight given to DO levels in invaded systems. Groups 3 and 4 show the most balanced attention across habitat attributes.

[24] Importance scores are directly affected by the range of attribute levels.

Table 2.7 Mean part-worths by expert group based on ratings

Attribute level	Sign. neg.	Mod. neg.	None	Mod. pos.	Don't know
	Expert rating of elodea's overall effect on salmonids				
Respondent count (n=56)	10	35	3	1	7
State of habitat					
Elodea-invaded	-156.85	-128.90	-90.22	-89.31	-119.62
Indigenous	156.85	128.90	90.22	89.31	119.62
Species					
Sockeye	7.73	8.90	2.60	35.16	9.13
Coho	12.85	9.87	-15.36	14.64	23.90
Chinook	-5.37	-12.80	-10.88	-19.77	-19.13
Dolly Varden	2.42	3.59	3.67	-5.21	-10.90
Whitefish	-17.63	-9.57	19.98	-24.82	-2.99
Location of vegetation					
Backwater	26.60	9.69	15.23	15.63	33.36
Entire system	-20.88	-9.61	-7.11	-51.42	-25.36
Lake	-5.72	-0.08	-8.12	35.78	-8.00
Vegetation cover [a]					
50% invaded	47.04	34.99	72.39	22.34	40.98
100% invaded	-47.04	-34.99	-72.39	-22.34	-40.98
0% indigenous	6.41	-1.25	-16.10	-7.84	7.93
50% indigenous	-6.41	1.25	16.10	7.84	-7.93
Dissolved oxygen (mg/l) [a]					
0.5 invaded	-49.27	-110.33	-152.82	-138.84	-83.85
10.5 invaded	49.27	110.33	152.82	138.84	83.85
5.5 indigenous	-56.37	-53.62	-74.34	-90.47	-27.90
10.5 indigenous	56.37	53.62	74.34	90.47	27.90
Prey abundance (mg/m^2) [a]					
30 invaded	-35.47	-36.83	-1.67	-101.46	-30.40
3000 invaded	35.47	36.83	1.67	101.46	30.40
400 indigenous	-13.58	-13.67	3.24	9.30	0.32
600 indigenous	13.58	13.67	-3.24	-9.30	-0.32
Piscivorous fish (#/acre) [a]					
20 invaded	17.68	16.65	13.62	9.81	12.12
35 invaded	-17.68	-16.65	-13.62	-9.81	-12.12
5 indigenous	52.94	50.25	-7.02	7.03	59.65
20 indigenous	-52.94	-50.25	7.02	-7.03	-59.65

a) Alternative-specific attribute level.

Interestingly, experts in group 5, who in the rating exercise stated to not know the overall effect of elodea on salmon, show preferences similar to experts in group 2. This result illustrates that these individuals were not necessarily outliers without expertise but may have been uncomfortable providing a rating even though their preferences show that they have substantial ecological knowledge consistent with other experts. Most importantly, this result shows the power and advantage of discrete choice methods over rating schemes, in providing a more user-friendly and accurate instrument for eliciting expert knowledge from a broader spectrum of experts, many of whom would be uncomfortable providing a rating. In the absence of the DCM approach, several of these experts may have opted out of the survey despite real expertise in the topic.

Table 2.8 Relative importance of attributes by group

		Expert rating of elodea's overall effect on salmonids				
		Sign. neg.	**Mod. neg.**	**None**	**Mod. pos.**	**Don't know**
Group ID		1	2	3	4	5
Respondent count		10 (18%)	35 (62%)	3 (5%)	1 (2%)	7 (13%)
Attribute						
Salmonid species		7.51	7.26	8.82	5.45	8.37
Location of vegetation		5.29	3.96	3.73	7.93	8.38
State of habitat		***28.52**	***23.44**	**16.40**	16.24	***21.75**
Vegetation cover	(invaded)	8.55	7.17	13.16	4.06	7.91
	(uninvaded)	7.97	4.36	3.03	1.43	5.01
Dissolved oxygen	(invaded)	8.96	**20.33**	***27.79**	***25.24**	**16.20**
	(uninvaded)	**10.25**	10.16	13.52	16.45	7.26
Prey abundance/m²	(invaded)	6.45	7.27	8.10	**18.45**	5.53
	(uninvaded)	3.14	3.02	1.26	1.69	5.06
Piscivorous fish/acre	(invaded)	3.75	3.84	2.91	1.78	3.71
	(uninvaded)	9.62	9.21	1.28	1.28	10.84

Note, in bold are the two most important attributes, with * indicating the most important attribute. The importance scores sum to 100.

Figure 2.2 illustrates that most experts do not believe salmonids can persist in invaded habitat with a smaller proportion detracting from this view. A closer look at how individual choice probabilities vary within and across groups reveals some inconsistencies between expert's choices and their ratings of the overall effect of elodea on salmon (Figure 2.6). Seven experts, who rated elodea as either significantly negative or moderately negative for salmonid persistence, show estimated choice probabilities that are much higher considering their rating (outliers in upper left of Figure 2.6). Additionally, the single expert in group 4 who rated elodea as moderately positive shows estimated choice probability for salmonid persistence in elodea-invaded habitat much lower than the sample overall. This result is contrary to what the expert stated in the rating task. The group, who collectively rated elodea as having no effect on salmonids, shows choice behavior that is consistent with their rating. Somewhat surprising, experts who did not know how to rate elodea's overall effect, have choice probabilities most representative of the sample overall, again supporting the advantages of choice methods in expanding the expert pool.

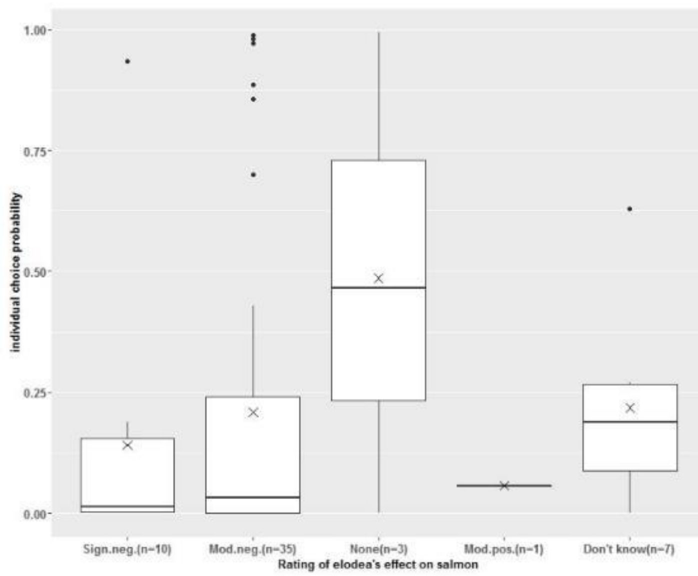

Figure 2.6 Boxplot of individual choice probabilities of salmonid persistence in elodea-invaded habitat by expert group. Lower and upper quartile (box), group median (bold line), group mean (x).

As noted earlier, the estimation of individual choice probabilities can be sensitive to the simulation method used. Figure 2.4 compares the 'randomized first choice' and 'share of preference' methods, showing insignificant differences in individual choice probabilities estimated by the two methods. As expected, the randomized first choice method results in more experts have choice probabilities closer to 0 and 1 compared to the share of preference method. The assumption that experts carefully evaluate habitat alternatives is supported by the many open-ended comments describing how experts evaluated the relationship between habitat attributes and supports the use of the randomized first choice method.

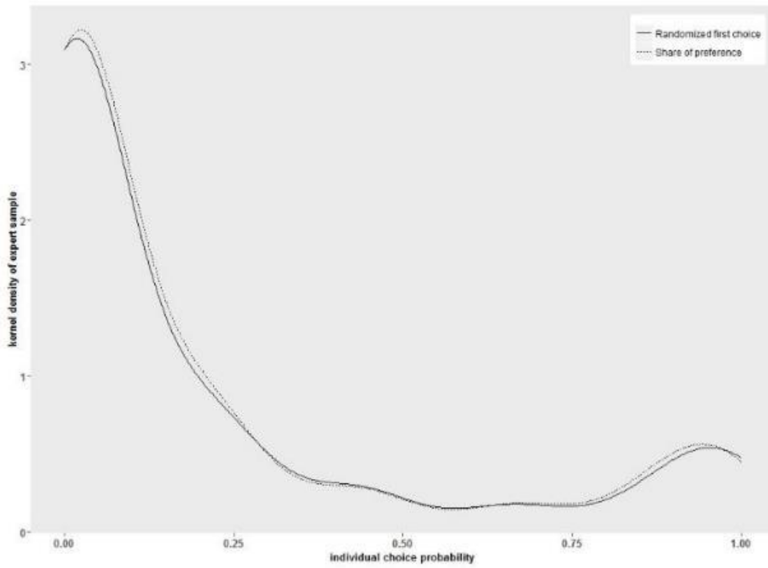

Figure 2.4 Kernel density of individual choice probabilities by simulation method.

2.3 Discussion

Existing risk assessment protocols rely on expert judgment, which often fail to separate 'current knowledge" from 'personal values' (Maguire 2004). For example, Alaska's invasiveness ranking system asks experts whether the species' potential to be spread by human activity is low (human dispersal is infrequent or inefficient), moderate (human dispersal occurs regularly), or high (there are numerous opportunities for dispersal to new areas) (Carlson, Lapina, and Shephard 2008). There are no clear quantitative definitions of what "infrequent", "inefficient", "regularly" or "numerous" mean in practice. Thus, the rating becomes a mix of judgments and personal definitions of the stated terms. As long as personal decisions depend on personal judgments, there is no problem. However, once judgment is provided on behalf of the public, the resulting ratings become prone to error as others may hold different definitions of the qualitative terms used in the assessment (Maguire 2004). Additionally, risk assessments tend to separate out the connected social values that are at stake, which could further inform the efficient allocation of resources to manage a list of "prioritized species", under limited budget. For example, the Alaska's invasiveness ranking system solely asks experts to rate ecological characteristics on a relative scale, failing to quantify and distinguish risk for invaders with potential catastrophic consequences versus invaders with just "bad" consequences.

Without being able to quantify risk in probabilistic terms (and quantify the quality of the assessment) large-scale management actions that require substantial public investments, may increasingly face public scrutiny. This study shows that DCM can not only help quantify the social values at stake (Hoehn, Lupi, and Kaplowitz 2010), but also quantify probabilities related to potential outcomes and as such serves two important elements for communicating risk to the public. In this context, DCM provides expert information to managers on whether resources should be deployed for taking action, serving information beyond species prioritization. In addition, DCM can provide critical information on the quality of an assessment and expands the set of available tools for resource managers making decisions on conservation investments (Maguire 2004; Turner and Daily 2007).

The study results have direct practical implications not only for statewide management of elodea in Alaska, but also for improved, evidence-based invasive species management. The establishment of elodea in Arctic and Subarctic locales illustrates the vulnerability of these regions to invasive species as new transportation corridors open (CAFF 2013; Heikkinen et al. 2009). DCM provides refined expert input on increasingly complex management challenges requiring action. In order to efficiently allocate society's resources, refining our understanding of trade-offs beyond relative risk assessment is critical for socially optimal decisions (Jason F Shogren 2000). On an agency level, investments to manage invasive species compete with an array of other management goals such as agricultural and wildlife management allocations. As a society, invasive species investments compete against broad social goals such as funding for children's health and education.

While this study should be considered a proof of concept with various advantages stated earlier, it is subject to several limitations, some of which could be addressed by changing the design and others should spur future research. First, this study chose persistence/extirpation as the system outcome used to develop a set of discrete choice sets for experts to consider as they examined the influence of multiple ecological factors (e.g., presence of elodea, DO, vegetation cover) on salmon. The specific discrete outcome – persistence/extirpation – represents a very clear but rather extreme ecological outcome. Managers have significant interest in less extreme outcomes associated with the potential effects of elodea on salmon. This issue may be addressed by altering the design to include an outcome variable in form of an attribute or by combining the discrete choice task with a rating exercise selecting from a scale or a best-worst scaling task

(Carson and Louviere 2015). While the advantages of such extensions are apparent, they come at the cost of additional respondent burden and may require additional skill that again limits the expert pool. An alternative experiment could also examine expert perspectives on changes in the abundance of salmon rather than the persistence/extirpation dichotomy.

Second, depending on specifics of the scenario, particularly when less dramatic outcomes are not presented, experts may differ in how critically they consider the idea that the change in ecosystem described in the scenario is resulting in complete loss of salmon. This problem may motivate more research into the best designs resulting in optimal applicability of DCM for eliciting indirect probabilistic knowledge and its use within decision analysis. Finally, an additional extension to the study could scale and then match the derived choice probabilities to probabilities assessed through biophysical experiments focused on estimating the effects of elodea on salmonids (Hensher, Rose, and Greene 2005). The original expert model can then be validated and updated (Drew and Perera 2011). An example, where the presented approach could serve as an alternative to directly encoding probabilities, is the probability assessment of extreme events particularly for population viability analysis (PVA) (Burgman et al. 2012).

Lastly, the marginal increase in research effort to develop more structured approaches to expert elicitation as presented here, may be viewed as not being as substantial as the savings potential associated with more informed decisions. As such, the DCM approach expands the tool box available to the expert elicitation practitioner.

3 Estimating market damages to commercial sockeye salmon fisheries: Integrating structured expert judgment with market valuation

3.1 Methods

3.1.1 Expert interviews

An extensive literature review of nearly 300 sources identified an expert pool of 111 individuals with substantive knowledge in three main areas: Pacific salmonids in freshwater habitat, the ecological role of submersed aquatic vegetation, and freshwater aquatic invasive plants. Experts were selected based on the number of citations in peer-reviewed publications using Google Scholar. Due to the localized issue of elodea in Alaska, the expert pool was expanded to include state and federal resource management agencies with job titles including fishery biologist, fisheries scientist, fish habitat biologist, and invasive species specialist. These individuals brought knowledge on localized variability and local observations to the expert pool.

In 2015, a mailed letter invited experts to participate in a scenario-based elicitation related to elodea's potential ecological effects on salmon persistence in elodea-invaded habitat. This elicitation quantified how expert opinion on persistent salmon populations was sensitive to varying assumptions on habitat and invasion characteristics (Chapter 1).[25] A follow-up questionnaire collected experts' judgment on the annual average growth rates expected for sockeye salmon in elodea-invaded habitat. Both the scenario-based elicitation and the follow-up included a four-page background document describing elodea's ecological effects (Supplemental file). This extensive background document and the structured question format of the scenario-based elicitation were aimed at reducing overconfidence in the follow-up interval judgment (Speirs-Bridge et al. 2010).[26] There the annual average growth rate was referred to as salmon production over many life cycles

[25] Environmental characteristics included location of elodea within the salmon system, description of the salmon system, dissolved oxygen levels, predation, prey abundance, and other factors.

[26] Even though the existing literature describes the reductions in overconfidence relating specifically to the 4-step interval elicitation procedure, the more elaborate nature of the scenario-based approach prior to the interval judgment is believed to have similar overconfidence-reducing effects. While a test of this assumption could be subject to future research, it is outside the scope of this study.

manifesting itself as a long-term “trend in abundance” (McElhany et al. 2000). Then the elicitation asked experts the following questions:

Q1. Imagine Alaska's sockeye salmon systems would be invaded with Elodea spp. and you had long-term population records with estimated growth rates for a random sample of 100 of these sockeye systems. What range of typical growth rates would you expect to see, that is, rates you would see about half of the time? Please specify in % and use a "-" (minus sign) for decline rates. lowest growth rate expected half the time (%) ________, highest growth rate expected half the time (%) ________.

Q2. For sockeye salmon, what growth rate would cause you to be concerned about extirpation of the population? Please specify in % and use a "-" (minus sign) for decline rates.

While the first question specifies the 25th and 75th percentile of the probability distribution related to the annual average growth rate in elodea-invaded sockeye salmon habitat, the second question is used to test expert’s comprehension of the task at hand and had nothing to do with elodea (Soll and Klayman 2004; Speirs-Bridge et al. 2010). This last question was used to eliminate experts who showed inconsistency by stating positive growth rates regarding extirpation concerns. Four experts stated positive growth rates in Q2 and were eliminated. Assuming normality in the stated growth rates, each expert’s interval judgments were used to form a normal distribution for that expert. The normality assumption is valid considering that many unknown ecological processes are likely at play in elodea-invaded habitat averaging out over a large sample. The combination of SEJ used equal weights (Morgan, 2014).

3.1.2 Commercial Fisheries Market Model

For the analysis of potential damages to commercial fisheries, Alaska is divided into five regions encompassing large watersheds that produce sockeye salmon and have commercial salmon fisheries associated with them (Figure 3.1) (USGS 2016). The regions are Bristol Bay and Kuskokwim in western Alaska, and Cook Inlet in Southcentral Alaska. The third region is Kodiak encompassing the island of Kodiak and southern coast of the Alaska Peninsula. The fifth region is called Gulf and includes the Gulf of Alaska coast of the Kenai Peninsula, Prince William Sound, and watersheds supporting the Copper and Bering River fishing districts. This region-specific analysis accounts for varying seafood processing capacity, run sizes, harvest levels, and prices.

The regions are also closely aligned with watershed boundaries chosen for further analysis extending this study. Salmon fisheries in Southeast Alaska were not part of this study, keeping the focus on regions of Alaska closest to the known elodea infestations.

Potential annual damages from elodea invasions to commercial fisheries are estimated over a hundred year period in discrete time. The model calculates the forgone net benefits to consumers resulting from a decrease (increase) in annual harvest and a consequential increase (decrease) in the price per lbs, assuming a linear and downward sloping demand function (Freeman 2003).

Since the SEJ-derived probability distribution entails positive and negative growth rates for salmon, this approach allows for potential positive and negative net changes in consumer surplus. These net changes imposed by quantity changes in annual harvest are then equal to the change in area under the ordinary (Marshallian) demand curve and equal to the consumer surplus in year t minus the consumer surplus in year $t+1$. In mathematical terms, annual damages per region are equal to

$$\begin{aligned}\Delta CS_{t+1} &= CS_t - CS_{t+1} \\ &= \frac{\gamma}{2}\left(h_t(p' - p_t) - h_{t+1}(p' - p_{t+1})\right)\end{aligned} \tag{3.1}$$

where γ is processing yield, h is sockeye harvest in lbs, p is the real (inflation-adjusted) per lbs wholesale price for sockeye salmon in 2015 USD received by Alaska primary processors. Prices are weighted by sockeye product ratios commonly observed in the Alaska processing sector. The choke price at which demand ceases is equal to p'. Using the own-price elasticity of demand, ε , the choke price equals $p' = h_t / \varepsilon + p_t$. Further, harvest in period $t+1$ can be expressed as a function of SEJ-derived growth rates θ , thus $h_{t+1} = f\left(h_t, \theta\right)$. After substituting and rearranging, Equation 3.1 becomes

$$\Delta CS_{t+1} = \frac{\gamma}{2}\left[\left(h_t - f(h_t,\theta)\right)\left(\left(\frac{h_t}{\varepsilon} + p_t\right) + \frac{f(h_t,\theta)p_t}{h_t\varepsilon}\right) - p_t h_t\right]. \tag{3.2}$$

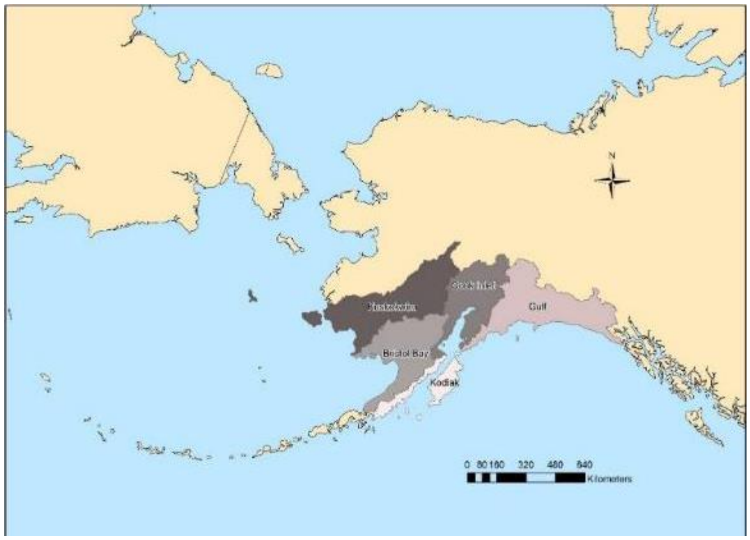

Figure 3.1 Major watersheds supporting Alaska commercial salmon fisheries targeting sockeye salmon.

Following this approach, several simplifying assumptions are made relating to the economic and environmental conditions of the commercial salmon fisheries with and without an invasion. First, the model assumes that harvest and abundance are proportional. Thus, a percentage change in abundance equals the same percentage change in harvest. Further, the predicted changes caused by elodea only serve to decrease (or increase) the weight of fish landed. Therefore, the predicted changes do not change the consumer demand function. Second, market conditions are assumed to be in equilibrium so that there are no incentives for harvesters and processors to enter or exit the market. Similarly, participation by fishing permit holders does not change over time. Third, wholesale prices are assumed to proxy prices for the end consumer. Analysis based on retail prices would be much more difficult, complicated by exchange rates and a multitude of retail products. In addition, retail prices are a reflection of other price factors non-attributable to salmon such as store location, parking, and availability of other products which result in significant variation in price (Knapp, Roheim, and Anderson 2007). Lastly, the fishery is assumed to be optimally and sustainably managed, ignoring various management inefficiencies. While over-capitalization remains an issue for Alaska salmon fisheries due to regulations resulting in a derby-style “race for fish” among other factors, there is evidence that the fisheries are sustainably managed (Steiner, Criddle, and Adkison 2011). Also, the high permit prices in many of Alaska’s salmon fisheries demonstrate that many fisheries create significant economic rent.

3.1.3 Model parameters

The correlation of prices among regions is modeled based on estimated Spearman's rank-order correlation coefficients observed between 2000 and 2015. The most widely produced product across regions, frozen headed and gutted, is used to determine this correlation. (Table 3.1).[27] The distribution of damages is analyzed using five Monte Carlo simulations each with 10,000 iterations (Palisade Corporation 2016).[28] The mean and standard deviations of real (inflation adjusted) wholesale prices are assumed to be lognormally distributed (Table 3.2).

Table 3.1 Spearman's rank-order correlation coefficients

	Bristol Bay	Cook Inlet	Kuskokwim	Gulf	Kodiak	Chignik a)
Bristol Bay	1.00					
Cook Inlet	0.89	1.00				
Kuskokwim	0.06	0.26	1.00			
Gulf	0.78	0.89	0.44	1.00		
Kodiak	0.80	0.90	0.18	0.73	1.00	
Chignik a)	0.80	0.90	0.18	0.73	1.00	1.00

a) Due to lack of data, assumes prices behave similarly to Kodiak. Note, correlations are based on just one product: frozen headed and gutted sockeye salmon.
Source: Alaska Department of Fish and Game Commercial Operators Annual Reports.

Harvest at $t=0$ is assumed to be lognormally distributed with the mean and standard deviation determined by harvest records observed between 2006 and 2015 (Table 3.2). Since the standard deviation compared to the mean harvest level in most regions is relatively small, the skew is small and thus the lognormal distribution tends to approach a normal distribution bounded by zero.[29] Since the return of salmon from different populations can vary within the same year, each harvest distribution is assumed to be independent of all others (Schindler et al. 2010). Note, while the model treats Chignik as a separate region because of available harvest data, the analysis of damages presented below will combine Kodiak and Chignik, consistent with the regional definitions mentioned above.

[27] Spearman's rank-ordered coefficients are more appropriate for modeling correlation among distributions compared to Pearson's correlation coefficients (Palisade Corporation 2016).

[28] Sampling type: Latin Hypercube, random number generator: Mersenne Twister.

[29] The distribution was constraint between zero and 1.5 times the maximum harvest to facilitate logistic growth assumptions presented below.

Using region-specific product shares observed among the processing industry (Table 3.3) and the distributional assumptions outlined above, a weighted wholesale price p_0 was calculated for each region. Processing yield γ is equal to the weighted average of product shares and sockeye salmon product-specific yields (Table 3.3) (Knapp, Roheim, and Anderson 2007).

Table 3.2 Commercial salmon fisheries characteristics by region

Region	sockeye harvest ('000 lbs)			Sockeye mean (SD) wholesale prices (real $/lbs) [a]			
	Mean	SD	Max	canned	frozen	fresh	other [b]
Bristol Bay	154,193	29,873	184,792	$ 3.66 (2.4)	$ 4.01 (2.3)	$ 2.71 (1.1)	$ 7.54 (2.5)
Cook Inlet	18,920	6,917	36,216	n/a [e]	$ 4.19 (3.0)	$ 3.40 (2.5)	$ 8.24 (6.3)
Gulf	16,386	5,173	24,785	$ 5.69 (2.9)	$ 3.79 (2.7)	$ 4.20 (2.4)	$ 6.30 (2.9)
Kodiak	11,980	3,109	17,007	n/a [e]	$ 3.22 (2.8)	$ 3.12 (1.3)	n/a
Chignik [c]	9,338	4,893	17,889	n/a	$ 3.22 (2.8)	n/a	n/a
Kuskokwim [d]	746	355	1,379	n/a	$ 1.11 (1.2)	n/a	n/a

a) Mean (standard deviation) in 2015 USD adjusted for inflation using the Anchorage Consumer Price Index. b) Salmon roe products Sujiko in Bristol Bay and mainly Ikura in Gulf. For Cook Inlet: fillets with skin no ribs. c) Assumes Kodiak prices due to lack of data. d) Prices reported for the exclusive economic zone (EEZ) were used due to lack of data. e) Region stopped production of this product or production is very inconsistent from year to year due to swings in run size. Source: Alaska Department of Fish and Game Fisheries Management Annual Reports and Commercial Operators Annual Reports.

Market-based approaches to valuation concerning changes in consumer surplus rely on measures that explain how responsive consumer demand for the marketed good (salmon) is to changes in the price of that good (Freeman 2003). Unfortunately, there are no specific estimates of these own-price elasticities for Alaska sockeye salmon. However, a distribution of own-price elasticities can be constructed from past studies for which elasticities vary widely (Table 3.4).

Table 3.3 Production shares by region

Product	Bristol Bay [a)]	Cook Inlet [b)]	Kuskokwim	Gulf [c)]	Kodiak [b)]	Chignik [a)]
Canned	0.32			0.34		
Fresh	0.02	0.12		0.08	0.12	
Frozen	0.64	0.86	1.00	0.57	0.88	
Other	0.02	0.02		0.01		1.00
Processing yield [d)], γ	0.70	0.78	0.75	0.71	0.78	0.75

a) McDowell Group (2015). b) Author estimates based on observed historic prices (ADFG 2016b; Knapp, Roheim, and Anderson 2007).c) Knapp, Roheim, and Anderson (2007). d) Weighted using product-specific yields: canned 0.59, fresh 0.97, frozen (headed & gutted) 0.75, other 0.75 (Knapp, Roheim, and Anderson 2007, author assumptions for other).

Following a market-based approach to valuation historical perspective on the global market for Alaska sockeye is helpful. Prices for wild Alaska salmon have been depressed in the 1990s with the onset of salmon farming resulting in rapid and sustained growth. Yet over the past decade, prices recovered due to marketing efforts aimed at wild and sustainably caught Alaska salmon, in addition to disease outbreaks in salmon farms elsewhere (Knapp, Roheim, and Anderson 2007).[30] In 2014, wild salmon comprised about 30% of global salmon production by volume. Of the wild salmon portion of this global market, Alaska sockeye salmon production took the largest share with 65% in wild sockeye volume of which 37% were caught in Bristol Bay (McDowell Group 2015). With the Bristol Bay sockeye salmon fishery being the world's most valuable wild salmon fishery, it can be argued that Alaska sockeye production plays a price-influencing role globally further supporting the approach (Knapp, Guettabi, and Goldsmith 2013).

There are several arguments that would support higher elasticities such as the existence of very close substitutes to wild sockeye salmon like coho, pink, or chum. Additionally, wild sockeye is considered a normal good where demand increases with rising income and vice versa. To the contrary, brand loyalty to a wild and sustainably harvested product is an argument for more inelastic demand if current marketing efforts and consumer awareness continue. For these reasons, the analysis uses the minimum (-12.78) and maximum (-1.472) elasticities for sockeye shown in Table 4.4 to bind a uniform distribution for simulation purposes. These distributional assumptions are grounded in observing comparable elasticities for other salmon (Table 3.4).

[30] Alaska's constitution prohibits salmon farming in state waters within 3 nautical miles.

Table 3.4 Own-price elasticities related to Pacific salmon wholesale prices

Salmon species	Market geography	Product	Mean elasticity	Source
Sockeye	Canada; Export	Canned	-9.67; -12.78	(DeVoretz 1982)
Sockeye	Canada	Canned	-2.141	(Wang 1976)
Sockeye	Pacific NW	Canned	-4.82	(Johnston and Wood 1974)
Coho	Oregon	Fresh/frozen	-9.68	(Swartz 1978)
Pink	Canada; Export	Canned	-12.92; -13.62	(DeVoretz 1982)
Chum	Canada; Export	Canned	-10.38; -2.90	(DeVoretz 1982)

The discount rate is another key uncertainty for which a triangular distribution was used assumed a range between 1% and 6% with a peak of 3%. This assumption is consistent with the real 30-year discount rate recommended by OMB and similar analysis of invasive species risk recently conducted (Rothlisberger et al. 2012; OMB 2016).

Table 3.5 Descriptive statistics for simulation inputs related to first wholesale value of sockeye salmon by region in millions of 2015 USD, base case (no invasion)

Region	Min	Mean	Median	5%	95%
Bristol Bay	77.4	427.4	387.4	191.7	804.0
Cook Inlet	4.8	61.5	49.1	17.5	148.7
Gulf	6.2	50.9	44.8	20.3	103.2
Kodiak	4.0	52.4	39.0	12.6	137.4
Kuskokwim [a)]	0.01	0.6	0.4	0.07	1.9
Total	123.0	592.5	533.8	268.8	1,107.0

The input assumptions to the simulation can be checked by comparing them with the observed historic first wholesale values.[31] For example, first wholesale values for Bristol Bay sockeye reached between $300 and $350 million six times in the early 1990s, which equals more than $700 million in 2015 USD. This historic value is within the 90% uncertainty range of the input distribution assumed for Bristol Bay (Table 3.5). Moreover, the model's median first wholesale value assumption for Bristol Bay is approximately equal to the observed 2010 first wholesale value further validating simulation input assumptions (Knapp, Guettabi, and Goldsmith 2013).

[31] First wholesale value is equal to the sum across each wholesale product's price times that product's sold volume and is equal to processors' gross earnings.

3.2 Expert judgment

A total of 56 experts responded to the invitation with 46 experts participating in the interval judgment. The remaining ten experts were unreachable or had retired by the time of the follow up. Two thirds of respondents were Alaska residents. The range of uncertainty varies substantially across individual experts (Figure 3.2). Ten experts (Group 1) expect sockeye to have positive mean annual average growth rates in elodea-invaded habitat.[32] Sixteen experts (Group 2) believe that on average elodea will not have an effect on salmon growth but confidence levels vary quite dramatically within this group. The majority of experts (18) believe that the mean growth rates to be expected is negative (Group 3). However, a third of this group recognizes that elodea can have positive growth effects on sockeye as shown by the extent of the 25th percentile above zero growth (Figure 3.2).

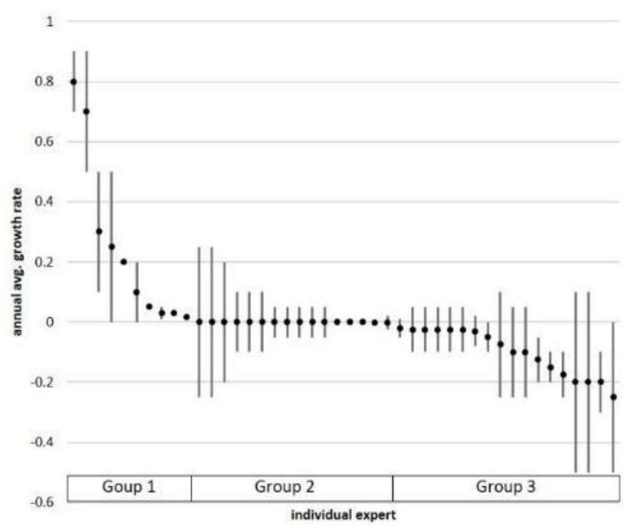

Figure 3.2 Stated annual average growth rate intervals (25th, mean, and 75th percentile) for sockeye salmon in elodea-invaded habitat by individual experts. Mean growth rates are stated to be positive (Group 1), zero (Group 2), and negative (Group 3).

[32] This believe is driven by elodea's ability to increase dissolved oxygen levels and to a lesser degree ecosystem productivity resulting in increased prey abundance, which these experts believe outweigh other more negative growth effects (elodea enhances pike habitat with pike being a salmon predator) (Chapter 1).

4 Estimating non-market damages to recreational floatplane pilots: An application of stated geographic preferences

4.1 Methods

4.1.1 Data collection instrument

A stratified random sample of 1,015 floatplane-certified pilots residing in Alaska was drawn using the publicly available Airmen Certification Releasable Database that includes the name and physical address of pilots and their certifications (FAA 2015).[33,34] The sample frame was divided into an urban and rural strata. While all 271 pilots residing in rural areas were included in the sample, a random sample of an additional 744 was drawn proportionally from the urban strata (Table 4.1).

Table 4.1 Stratified random sample

Strata	Population of floatplane pilots	%	Sample	%
Municipality of Anchorage, Cities of Palmer and Wasilla	1,733	66%	548	54%
Kenai Peninsula Borough	227	9%	72	7%
Cities of Fairbanks and North Pole	342	13%	108	11%
City of Kodiak	52	2%	16	2%
Urban total	2,354	90%	744	73%
Rural total	271	10%	271	27%
Total	2,625		1,015	

The survey was conducted between December 2015 and May 2016, without a non-response survey, thus preventing analysis of selection bias. First, pilots were contacted using a

[33] According to the FAA, opt-out rates for not wanting to release personal data in a public database are minimal.

[34] Southeast Alaska was excluded for several reasons. Floatplane bases are almost exclusively in saltwater, minimizing risk of freshwater invasive species transfer. So far, aquatic invasive species have not been found in Southeast Alaska. Only 8% of Alaska's floatplane-rated pilots reside in Southeast Alaska.

letter of invitation with survey URL and $2 incentive payment, followed by a post card reminder, and finally a third reminder including a hard copy survey with paid mail return option (Dillman 2007). The hard copy allowed respondents with no or slow Internet to respond. As of 2014, 87% of Alaska households had Internet access (U.S. Census Bureau 2014). There is no reason to believe that those pilots without Internet would have had different floatplane flying behavior than those with Internet. Data collection also included a telephone interview with web survey for 100 commercial floatplane operators in the state and pilots who fly for government agencies.

The web survey contained three parts, beginning with questions about awareness of aquatic invasive plants and an introduction to the floatplane safety hazards related to elodea. The second and main part included an electronic mapping tool (Figure 4.1), followed by final questions related to floatplane operating costs and brief questions on personal demographics. The online mapping tool allows pilots to precisely identify their flying destinations and avoid spatial ambiguity. Respondents were asked about their home base and then asked to mark their 2015 first-leg freshwater flight destinations.[35] Key informant interviews showed that most pilots fly to a destination then return to their home bases with few flights containing more than one destination after take-off from a home base. It should be noted that the risk of invasive species dispersal is greatest for the first destination after take-off from urban source locations.

For each destination, pilots were further asked via a small pop-up window to state the approximate number of trips flown to the marked location in 2015. Then a hypothetical question asked respondents to specify the anticipated number of annual flights given the landing zone would be covered in dense aquatic vegetation (Figure 4.1). This question in particular collected both pre and post invasion flight behavior and thus creates a panel dataset essential for valuing damages (Hausman, Leonard, and McFadden 1995). Potential problems with this approach relate to respondents' difficulty of remembering their recreational activity from a year ago, and the hypothetical nature of the stated behavior. The development of the survey instrument was particularly conscious of respondent burden and thus kept the mapping exercise as simple as possible. This approach had obvious trade-offs as it prevented the instrument from becoming

[35] Key informant interviews showed that most pilots fly to a destination then returning to their home bases with few flights containing more than one destination after take-off from a home base. Also, the risk of invasive species dispersal is greatest for the first destination after take-off from an urban source location, assuming potential aquatic vegetation drops off while landing (Hollander 2015a).

more complex - complexity that could have enabled the collection of more detailed information about their destinations, decision making, and flying behavior.

Figure 4.1 Online mapping tool for eliciting floatplane destinations

Even though the survey instrument did not specifically ask for a second best destination, assuming the identified site would be invaded, the approach is still able to account for substitution. In specific, a downward shift in demand due to an invasion is partially offset by upward shifts in demand at other destinations. Key informant interviews with pilots who took the survey showed that pilots consider continuing to land in destinations with larger water depth because elodea invasions would predominately occur in shallower parts of a lake or waterbody. Pilots also mentioned that they would reduce or eliminate flying to destinations with shallower water depth, as these destinations would be more hazardous to land in, given the plants can cover the entire water surface.

Alternative approaches to data collection are more involved such as diaries (Carson, Hanemann, and Wegge 2009) or discrete choice experiments (Hanley, Wright, and Koop 2002). With the diary approach, data collection relies on the recording of recreation activity for a distinct pre and post invasion period. This is often expensive and time consuming, preventing timely data for management purposes. In addition, such approaches require often unknown information on the invasion status of the destinations to distinguish invaded from un-invaded recreation sites.[36] Lastly, despite the obvious advantage of the ability for discrete choice experiments to predict substitution patterns, they are more complex. This complexity would result in considerable response burden, particularly for pilots who fly to many destinations. Since the survey's main purpose was not to estimate non-market damages but instead to identify remote floatplane destinations, the decision was made to keep the survey instrument as short and as user-friendly as possible.

4.1.2 Recreational demand model

The pilot's decision is assumed to be two-fold. First, the pilot chooses an alternative site or chooses not to fly. The pilot then decides about the number of annual trips to the selected alternative. By using the number of flights as a frequency weight in the estimation process, the decision is reduced to one level.[37] The econometric specification of this decision is motivated by random utility maximization (RUM) (Manski and McFadden 1981). It defines overall utility of an alternative j to individual n as U_{nj}, comprised of observable utility V_{nj} and the unobservable utility, ε_{nj} thus $U_{nj} = V_{nj} + \varepsilon_{nj}$ (McFadden 1973). The measured component of utility in linear form for individual n is

$$V_j = \beta_{0j} + \beta_{1j} X_{1j} + \beta_{2j} X_{2j} + ... + \beta_{kj} X_{kj} + \beta_c C_j \tag{4.1}$$

where β_0 represents the average of all the unobserved sources of utility, $\beta_{1,...,k}$ is the coefficient that estimates the contribution of attribute $X_{1,...,k}$ to the observed sources of relative utility where X_1 is the first attribute in k number of attributes, β_c is the coefficient for the cost of the alternative and C is the travel cost attribute. The choice rule states that each individual evaluates all

[36] Since only 16 sites are currently known to have been invaded by elodea in Alaska, only ten of which are accessible via floatplane, this approach is unrealistic due to the small sample size.

[37] One could argue that there is a third level - flight distance. Respondents indicated small sets of destinations, often falling within one region leaving little variation in distances related to alternative site choices (Table 3.4).

alternatives presented, U_j for $j = 1, ..., J$ alternatives in the choice set, then compares $U_1, U_2, ..., U_J$ and finally chooses the alternative with maximum utility.

Common RUM are the multinomial logit (MNL) and multinomial probit (MNP) models (Chen, Lupi, and Hoehn 1997; Hausman, Leonard, and McFadden 1995; McFadden 1973; Hausman and Wise 1978). Both MNL and MNP are appropriate for modeling recreation demand because they are models of discrete choice that accounts for substitution among alternatives. The main difference between the MNL and MNP models lies in the distribution assumption of the unobservable part of utility, ε_{nj}, which leads to different assumptions regarding decision makers' substitution patterns.

In the MNL, the error term is assumed to be independent and identically distributed following a type 1 extreme value distribution. This assumption allows the choice probabilities to be easily calculated. It assumes that the ratio of probabilities of any two alternatives cannot change if any other alternative is added or taken away from the set of alternatives in a choice set. Put differently, the pattern of substitution is limited by the Independence of Irrelevant Alternatives (IIA) property. It assumes that a change in one alternative has the same effect on all other alternatives. Thus, all alternatives are assumed to be equally dissimilar with none being more or less similar to each other (Hausman, Leonard, and McFadden 1995). As such, the choice probabilities related to alternatives that pilot n chooses alternative, i, from a set of J alternatives presented in a choice set equals the multinomial logit (MNL) specification:

$$p_{ni} = \frac{e^{\beta_n X_{ni}}}{\sum_{j=1}^{J} e^{\beta_n X_{nj}}} \tag{4.2}$$

where $i \neq j$ and $j \in j = 1, ..., J$ (McFadden 1973). The probability ratio between two choices, h and g, is independent of the utility functions other than those for alternative h and g, or in mathematical form $\frac{p_h}{p_g} = \frac{e^{e^{\beta_n X_{nh}}}}{e^{\beta_n X_{ng}}}$, also known as the IIA property of the logit rule. In other words, the relative odds of choosing h over g are independent of all other alternatives and do not change with a new alternative, the result being proportional substitution. Since the IIA assumption can assume a rather unrealistic substitution pattern, the application of MNL to estimate recreational damages deserves care.

The MNP forms a nice alternative approach to remedy the IIA assumption. In the MNP, the random component of utility, ε_{nj}, is assumed to be correlated across choices and to be following a multivariate normal distribution. The resulting choice probabilities are

$$p_{ni} = \int_{-\infty}^{\infty} F_j \left((X_j - X_i)\beta_n + \varepsilon_{nj} \right) d\varepsilon_{nj} \tag{4.3}$$

where F_j is the joint distribution of the errors. Since the above integral is not closed and multi-dimensional, estimation of the choice probabilities relies on Monte Carlo simulation techniques such as Gibbs sampling. In contrast to the logit model, the probability ratio of the MNP depends not only on the utility functions for alternatives g and h but all alternatives, thus relaxing the IIA assumption (Chen, Lupi, and Hoehn 1997).

The welfare changes estimated from either of the two recreation demand models are equal to the total derivative of the utility function (Equation 4.1) with respect to changes in attribute X_k and C . This equals the change in cost that keeps utility unchanged given a change in X_k or in mathematical form:

$$\frac{dC}{dX_k} = WTP_k = -\frac{\beta_k}{\beta_C} \tag{4.4}$$

4.1.3 Approach

Particular care was given to the way the data was formatted for empirical analysis to allow substitution patterns to emerge and proper damage assessment to occur (Hausman, Leonard, and McFadden 1995). The data was coded as a panel where each panel refers to a choice set, one related to pre-invasion and the other to the stated post-invasion flight choices. Each respondent’s individual destinations were grouped into eight alternatives encompassing large watersheds defined by the National Hydrographic Dataset (NHD) (Figure 4.2) (USGS 2016). This aggregation was necessary for two reasons. First, the data showed more than 700 individual destinations, a number quite large for estimation purposes.[38] Second, the alternatives closely align with watershed boundaries set for further analysis extending this study to include damages to Alaska's commercial salmon fisheries. Thus the alternatives show the extent of watersheds supporting these fisheries. A no-fly ninth alternative was added to account for the flights less taken given elodea outbreaks. A binary choice variable indicates to which alternative the pilot flew.

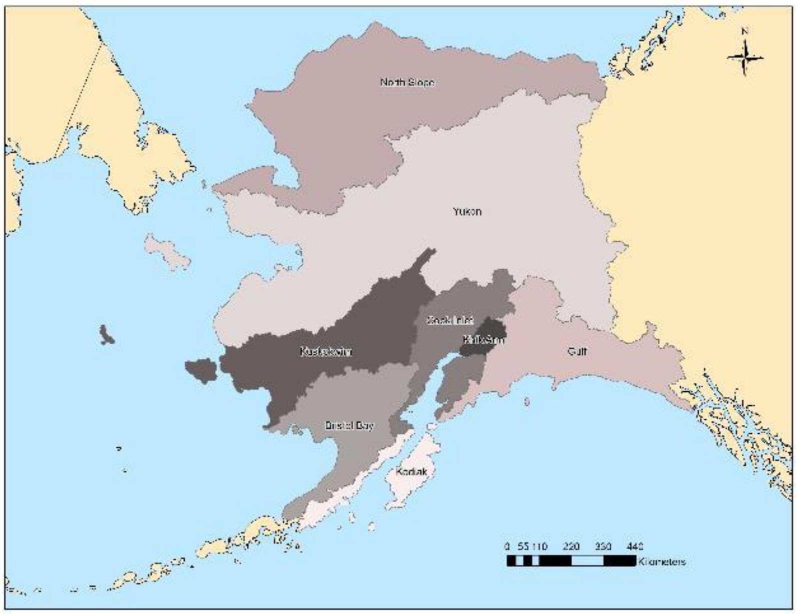

Figure 4.2 The eight regions that form alternatives in the recreation demand model.

Estimating the recreation demand model requires information on varying destination attributes to explain the choice of recreation destinations. Since the survey instrument did not ask about site characteristics or the motivation of pilots, the approach relies on external statewide data sources to describe how alternative destinations vary. Publicly available data on moose and sheep hunting success by game management unit[39] was used to assign destination attributes X_j (Alaska Department of Fish and Game 2016). The level of X_j was set to equal the mean kill to hunter ratio observed for each individual pilot's destinations within j. While hunting success ratios are a good indicator of hunting and wildlife viewing quality and one motivating factor in

[38] Many econometric software packages limit the number of alternatives in the choice model.

[39] The Alaska Department of Fish and Game divides Alaska into game management units (GMU) for the purpose of managing game species. These GMUs are aligned with watershed boundaries and are therefore geographic subsets of each region *j*.

pilots' decision to fly, there are obviously many more underlying unknown motivational drivers the model is unable to capture due to data limitations.

The remaining explanatory attributes include pilot age and travel cost. The cost to fly to each alternative region is individual-specific for regions the pilot chose to fly to and estimated for regions the pilot chose not to fly to. The stated plane operating cost, pilot's plane type and cruising speed (Table 4.4) were used to calculate a per km cost for each respondent multiplied by the weighted average of each respondent's Euclidean distances between home base and destinations within region j. Costs associated with alternatives to which the pilot did not fly, were estimated using the pilot's per km cost multiplied by the Euclidean distance between the home base and centroid of the destination region not chosen.

Non-participation in the survey is assumed to be randomly distributed across the population of pilots and was addressed via weighting. A frequency weight was calculated for every destination region chosen by the respondent equal to the stated flight frequencies scaled to the population of pilots in each strata as defined by the sample frame.[40] Since the sample frame does not include information on age, this prevents an adjustment for age. Using flight frequencies as a weight avoids having to model flight allocations among regions and simplifies the econometric estimation approach (Hausman, Leonard, and McFadden 1995).

4.2 Survey results

4.2.1 Survey response

Of the 1,015 initial mailings, fifteen were undeliverable. A total of 444 pilots responded for a response rate of 44%.[41] This relatively high response rate for a mailed invitation to participate in the web-survey may partly be due to heightened awareness of the problem of aquatic invasive species among floatplane pilots. In addition, despite the average respondent taking 24 minutes to complete the survey, response burden is adequate considering respondents' awareness of the problem. A total of 239 pilots report that they flew a floatplane in Alaska in 2015 but only 229 of

[40] Accounts for the observed proportion of pilots reporting that they did not fly floatplanes in 2015 for each of the strata.

[41] Includes 162 hard copy mail returns.

those provide mapping responses useful for analysis. Of the total respondents, 219 indicated to not have flown in 2015, and four respondents did not answer whether or not they flew (Table 4.2). A total of 114 pilots were willing to volunteer regarding monitoring and raising awareness among pilot circles. Responses from rural areas were proportionally larger, likely due to the oversampling in rural areas at the expense of undersampling in Anchorage, Wasilla, and Palmer. Responses from other urban areas were relatively proportional (Table 4.2). A total of 71 commercial operators responded as well as 25 government pilots.

Table 4.2 Response by strata

Strate	Respondent count	%	Map-response	%	Did not fly	%
Municipality of Anchorage, Cities of Palmer and Wasilla	209	47%	127	56%	95	43%
Kenai Peninsula Borough	35	8%	15	4%	19	9%
Cities of Fairbanks and North Pole	64	14%	33	14%	27	12%
City of Kodiak	6	1%	2	1%	4	2%
Urban total	314	71%	177	75%	145	66%
Rural total	130	27%	44	25%	74	34%
Total	444		221		219	

a) Excludes 233 pilots residing in Southeast Alaska.

Respondent characteristics are as expected. Half of the respondents were retirement age. Respondents' median personal income before taxes in 2015 is $135,000 compared to the most recent statewide median annual earnings of $30,800 (U.S. Census Bureau 2014). Pilots varied most by the number of annual flights they took in 2015 (Table 4.3). On average pilots reported taking between 30 and 40 flights over a roughly 100 day season with totals ranging from five to over 500 flights.[42] The average respondent's longest flight was 257 km, which is considerably less than the effective aircraft ranges (Table 4.4). On average, pilots carry one to two passengers, and the annual average number of unique destinations they fly to from their home base is between four and five. This small set of destinations suggests that most pilots primarily fly to a small number of preferred destinations where they are familiar with local conditions instead of engaging

[42] Key informant interviews showed that flying dates depend on different break-up and ice-up conditions across the state. Due to warmer temperatures observed in recent decades, the season has lengthened and in Anchorage has been 112 days long on average from June 1 to September 20 (Rust's Flying Service personal communication).

in more exploratory flying. This result also suggests that substitution patterns for pilots are much more limited compared to other modes of transportation involving less required experience and less risk-taking.

The estimated plane operating costs range from $0.10/km to $2.97/km with a mean of $0.83/km (Table 4.5). The most frequently flown aircraft among respondents is the Cessna followed by some of the smaller fixed wing single engine airplanes like the Piper Super Cub or Tailorcraft. A third of all respondents either did not specify an aircraft or selected the "other" category (Table 4.4).

Table 4.3 Respondent characteristics

	Personal income [a)]	2015 avg. # passengers	2015 flights	Pilot age	Number of unique destinations	Longest annual flight from home base(km)	Operating Cost ($/km) [b)]
Mean	$ 137,786	1.41	36	58	4.23	257	$ 0.83
Median	$ 135,846	1.00	25	58	3	222	$ 0.75
Mode	$ 135,846	1.00	5	58	1	185	$ 0.78
SD	$ 70,101	1.13	46	11	5	162	$ 0.51
Minimum	$ 25,000	0	5	26	1	3	$ 0.10
Maximum	$ 300,000	6.00	509	94	55	1,000	$ 2.97
Respondent count	229	229	229	229	229	213	229

a) Before taxes. b) Estimated based on cruising speed of plane type and stated operating cost.

Through geoprocessing of the destinations identified by respondents, characteristics of the destination waterbodies could be identified. An important criteria for access via floatplanes is fetch - the maximum uninterrupted water distance between any two points on the perimeter of a waterbody as defined by the NHD and excluding glaciers (USGS 2016). The fetch serves as a proxy for accessibility and can be used to calculate a subset of floatplane accessible waterbodies from the NHD. The minimum fetch of waterbodies to which respondents flew was estimated at 336 meters (Hollister 2016) (Table 4.4).[43] Floatplane accessibility based on the fetch criteria differs among regions. Statewide, about 16% of all waterbodies are accessible by floatplane,

[43] One has to recognize that the pilot's decision to land or not to land on a waterbody is much more complex than fetch, although it is one of the most important features. Additional decision factors include the pilot's flying experience and skill, topography such as tree cover near the lake shore and the surrounding terrain features combined with weather conditions and aircraft type.

given the criteria. The highest proportion of accessible waterbodies, almost half, lies in the Knik Arm region followed by the Gulf and Cook Inlet regions (both 20%). The North Slope and Kuskokwim have similar accessibility (17 and 18%). The Bristol Bay and Kodiak have the lowest accessibility with only 10% of waterbodies larger than 336 meters fetch (Table 4.5).

Table 4.4 Floatplane characteristics

Type of single engine plane	Respondent count	Passenger seats	Cruising speed [km/h]	Range [km]	Minimum fetch of destination [a)]
Piper PA-17, PA-18, Tailorcraft	49	1	163	493	336 m
Cessna-172 to 206	91	4	253	1325	511 m
DeHavilland DHC-2 Beaver	3	6	230	732	505 m
DeHavilland DHC-3 Otter	0	10	195	1520	645 m
Other and not specified	86	4	216	1030	498 m
Count	229				

a) Minimum fetch of stated destination waterbodies by airplane type in meters. Fetch is measured by the maximum distance between two points on the perimeter of the waterbody and was estimated using R (Hollister 2016).

The survey identified the most floatplane destinations (205) in the Yukon region, mainly due to this region being the one with the largest land mass. Cook Inlet which is closest to Anchorage where the highest proportion of floatplane pilots reside also shows a high destination count (187) (Table 4.5). Comparing the count of identified destinations to the subsets of accessible waterbodies based on fetch (column 4 Table 4.5) illustrates the extent to which pilots use each region and reveals the proportion of potential unknown destinations. In this context use rates in Knik Arm, Kodiak and Cook Inlet are higher than other regions. Lowest utilization rates occur in the Kuskokwim, due to the Alaska Range being in the way from Anchorage, and the North Slope due to the largest distances from urban centers. The density of accessible waterbodies (column 7 Table 4.5) is another characteristic in which regions vary, an index for accessibility and the degree to which pilots would be able to substitute between sites given elodea invasions in their destinations. In this context, the highest densities occur on the North Slope and in the Yukon regions, where there is more than one accessible waterbody per km^2 and one every two km^2 respectively (Table 4.5).

Table 4.5 Region characteristics

Region	Waterbody total count a)	Floatplane accessible b)	Survey count e)	Survey % of accessible c)	Region size in km²	Accessible/ km²)
Gulf	51,597	10,510	71	0.7 %	54,366	0.19
Knik Arm	2,019	979	28	2.9 %	12,629	0.08
Cook Inlet	38,165	7,707	187	2.4 %	53,375	0.14
Kodiak	15,271	1,471	41	2.8 %	44,028	0.03
Bristol Bay	126,394	13,086	74	0.6 %	53,297	0.25
Kuskokwim	182,194	30,576	28	0.1 %	75,744	0.40
North Slope	238,274	43,707	93	0.2 %	34,444	1.27
Yukon	360,549	58,033	205	0.4 %	115,544	0.50
Total	1,014,463	166,069	727		443,428	0.37

a) Waterbody count excluding glaciers (USGS 2016). b) Subset of total count based on floatplane accessibility determined from survey results to equal minimum fetch of 336 m. c) Fourth divided by third column. d) Third divided by sixth column. e) Number of waterbodies respondents identified they landed in.

Table 4.6 Mean attribute levels by alternative region

Region	Sheep (Success per hunter) a)	Moose (Success per hunter) a)	Cost ($) b)
Gulf	0.24	0.28	474
Knik Arm	0.19	0.16	356
Cook Inlet	0.23	0.19	212
Kodiak	0.00	0.37	1,029
Bristol Bay	0.00	0.28	685
Kuskokwim	0.64	0.59	453
North Slope	0.09	0.39	1,238
Yukon	0.21	0.32	655
Total	0.18	0.29	567

a) Successful kill per hunter for 2015, varies by game management within region. b) One-leg flight cost between home base and respondent destination. Varies by respondent and aircraft type.

To put the estimated non-market damages into perspective, the destination regions are further described by the observed attribute levels for moose and sheep hunting success as well as flight costs (Table 4.6). The Kuskokwim region has the highest success rates for sheep hunting while the Gulf region has the highest for moose hunting followed by the Yukon and Kuskokwim regions. Flying costs are highest to the North Slope followed by Kodiak, Bristol Bay and the Yukon

(Table 4.6). Table 4.7 illustrates the weighted number of flights survey respondents indicated to have flown from each region's busiest floatplane hub. These flight frequencies are later used in Chapter 5 to model spatial spread of elodea through floatplanes. About 57% of all floatplane flights to freshwater destinations originate from the primary floatplane hubs in each region identified by the highest number of flight operations (Table 4.7). Flight patterns from these hubs are fairly representative of all floatplane bases in each region. Deviation from the flight patterns observed in hubs occur in Cook Inlet where Lake Hood represents 10% fewer flights to Bristol Bay compared to other Cook Inlet bases. Additionally, Cordova's Eyak Lake represents 8% fewer flights to Cook Inlet compared to other bases in the Gulf region, and Bettles Float Pond represents 26% more flights to the Yukon compared to other bases in the North Slope region.

Table 4.7 Floatplane pathway between regional freshwater floatplane hubs and freshwater destinations in seven regions [a)]

		Number of 2015 flights to region of destination, v^i							
Regional floatplane hub	**Region of Origin**	**Bristol Bay**	**Cook Inlet**	**Kusko kwim**	**Gulf**	**Kodiak**	**North Slope**	**Yukon**	**Total**
Shannon's Pond, Dillingham	Bristol Bay	3,450	17	280	0	0	0	0	3,747
Lake Hood, Anchorage	Cook Inlet	1,903	25,382	105	580	0	0	206	28,176[b)]
Hangar Lake, Bethel	Kuskokwim	117	0	170	0	0	0	79	366
Eyak Lake, Cordova	Gulf	0	0	0	463	58	0	0	521
Lilly Lake, Kodiak[c)]	Kodiak	0	34	0	0	2,934	0	0	2,968
Float Pond, Bettles	North Slope	0	0	0	0	0	458	1,624	2,082
Float Pond, Fairbanks	Yukon	19	112	27	36	0	944	5,812	6,950
Other		10,497	13,161	583	2,411	2,939	786	3,903	34,280
Total		15,986	38,706	1,165	3,490	5,931	2,188	11,624	79,090

a) Weighted flight estimates based on survey responses observed in each strata for private pilots, all commercial operators, and government pilots. b) The only available data to validate this estimate are FAA operation counts at Lake Hood. However, the FAA does not distinguish flights based on landing gear, preventing a count of floatplane flights. Total Lake Hood operations count during open water between June 1 and September 1 was 35,140 (FAA 2016). The FAA count includes flights that are immediate returns to Lake Hood after take-off (without a destination) and counts flights to saltwater destinations. Therefore, the estimate of 28,176 floatplane flights to freshwater destinations is reasonable. c) The most frequent freshwater floatplane hub on Kodiak Island, many are are in saltwater.

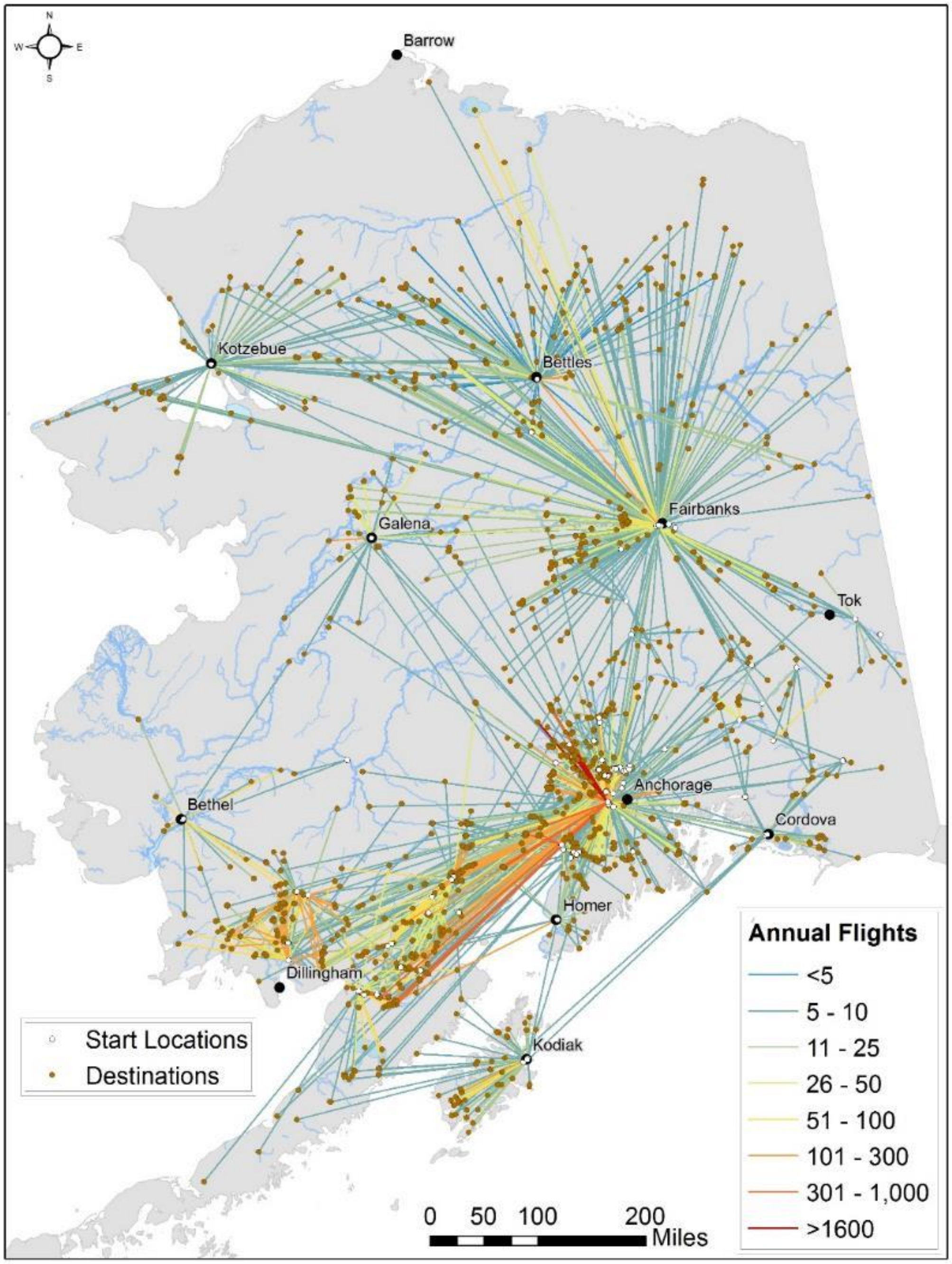

Figure 4.2 2015 flight patterns as stated by all floatplane pilots surveyed

4.2.2 Potential impact of elodea on pilots' flying behavior

Half of the respondents stated that they would no longer fly to destinations they flew to in 2015 if dense aquatic vegetation would be in the landing zone (Table 4.8). This result is supported by the Lake Hood floatplane base's aquatic management plan, stating safety concerns as the primary reason for continued aquatic vegetation management (CH2MHILL 2005). The survey also asked respondents about their knowledge of elodea and informed pilots about elodea's potential effects to pilot safety. About 75% of respondents had heard about elodea and reported safety concerns flying to destinations that are shallow and already required caution. Follow-up interviews with respondents indicate that pilots identified destinations by taking into account individual lake characteristics such as water depth and terrain features. In addition, weather conditions, pilot skills, and plane models are significant drivers determining access (Table 4.8).

Table 4.8 Recreational pilots' stated change in flight behavior due to elodea-invaded destinations (n=229)

	Continue flying			Stop flying
	to all their destinations		only to some destinations	
	without flight reduction	with flight reductions	with flight reductions	
Pilot count (%)	43 (19%)	36 (16%)	35 (15%)	115 (50%)
Mean % change in annual flights	0%	-40%	-58%	-100%

4.2.3 Pilot user loss estimates

Multinomial logit and multinomial probit models were fit using maximum likelihood optimization.[44] The coefficients by attribute are shown in Table 4.9 and Table 4.10 for the MNL and MNP models respectively. All coefficients have the expected signs and are statistically significant at the 0.001 level. For calculating the estimated loss in floatplane pilot use values from elodea invasions, statistical significance of the elodea invasion and cost coefficients is of particular importance. This empirical result is supported by more than three quarters of respondents indicating that they had heard about the spread of elodea in Alaska and were aware

[44]STATA's generalized linear model command (glm of the binomial family with the logit or probit links) was used. This specification results in identical parameter values as using the mlogit or mprobit STATA commands.

of the risk it poses to floatplane safety. Not surprising are the coefficients for moose hunting and sheep hunting success, considering that Alaska has the highest participation rate in wildlife-related recreation by state residents in the U.S. (U.S. Fish and Wildlife Service and U.S. Census Bureau 2013). The positive coefficient on the age attribute is expected and reflects that flying is an expensive hobby which is largely enjoyed by those who are retired and have the disposable income needed to pursue the activity. White's robust standard errors are used to make valid statistical inference as data collection possibly caused the explanatory attributes and the error term to not be identically distributed as assumed by the model (White 1980).

Table 4.9 Estimated coefficients applying the MNL for explaining choice of alternative

Attribute	Coefficient	Robust Standard Error	z	p>|z|	95% Confidence Interval	
Elodea invasion	-0.296	0.021	-14.260	0.000	-0.337	-0.256
Cost	-0.002	0.000	-27.770	0.000	-0.002	-0.002
Moose hunting success	1.431	0.127	11.300	0.000	1.183	1.679
Sheep hunting success	2.270	0.078	29.010	0.000	2.117	2.424
Age	0.010	0.001	14.880	0.000	0.009	0.011
Constant	0.398	0.047	8.470	0.000	0.306	0.490
AIC (deviation)	1.0852					
Log ps likelihood	-53109					

Specifying the model as a multinomial probit model results in slightly more information being captured by the model as indicated by the lower AIC (Table 4.9) (Akaike 1974). The coefficients are comparable to the multinomial logit model in sign and magnitude with similar high precision. This similarity may suggest that the IIA assumption has little consequence as long as sufficient data quality minimizes the amount of unobserved heterogeneity (Hensher, Rose, and Greene 2005). This result is also supported by the consumer surplus changes estimated using both models (Table 4.11).

For the damage assessment, willingness to pay estimates and 95% confidence intervals are established using the Krinsky and Robb method with 2000 replications, which is a parametric bootstrap method that assumes that the coefficients in Equation 4.1 are normally distributed (Hole 2007; Krinsky and Robb 1986). Table 4.11 presents the marginal user loss per flight related to a floatplane destination being elodea-invaded. In order to validate the model used for the elodea damage assessment, additional WTP was calculated for moose hunting and sheep hunting success. In the discussion below, these estimates are compared to two studies containing wildlife-related WTP estimates in Alaska supporting validity of the model (Table 4.11).

Table 4.10 Estimated coefficients applying the MNP for explaining choice of alternative

Attribute	Coefficient	Robust Standard Error	z	p>|z|	95% Confidence Interval	
Elodea invasion	-0.183	0.012	-15.290	0.000	-0.206	-0.159
Cost	-0.001	0.000	-31.100	0.000	-0.001	-0.001
Moose hunting success	0.836	0.072	11.630	0.000	0.695	0.977
Sheep hunting success	1.279	0.045	28.160	0.000	1.190	1.369
Age	0.006	0.000	14.700	0.000	0.005	0.007
Constant	0.266	0.028	9.440	0.000	0.211	0.321
AIC (deviation)	1.0850					
Log ps likelihood	-53096					

Table 4.11 Estimated change in consumer surplus per flight

	MNL		MNP	
Attribute	**Mean**	**95% C.I.**	**Mean**	**95% C.I.**
Elodea invasion	-$178	-$205, -$151	-$185	-$157, -$211
Moose hunting success	$861	$736, $981	$848	$726, $965
Sheep hunting success	$1366	$1215, $1531	$1298	$1162, $1447

Assuming all destinations within a region are elodea-invaded, the per-flight consumer surplus estimates from Table 4.11 can be used to calculate potential floatplane use losses conditional on the number of flights to a region. This estimate is applied in Chapter 5.

5 A risk and decision analysis for managing invasive elodea: Linking pathway dynamics with changes to ecosystem services

5.1 Methods

Potential damages to commercial sockeye salmon fisheries were estimated in Chapter 2 following a market-based approach to valuation (Freeman 2003) that has previously been applied to fisheries in the invasive species context using structured expert judgment (Rothlisberger et al. 2012; Cooke 1991). The recreational user loss accruing to floatplane pilots was measured in Chapter 3 using the previously mentioned survey data from pilots applied to a recreation demand model (Hausman and Wise 1978). Here, both of these economic valuation studies are integrated to form a spatially and temporally explicit risk analysis that forecasts potential future damages and informs resource managers about optimal decision-making (Holmes et al. 2010). Below, the metapopulation model is first described and then extended to incorporate management action. The second part integrates ecosystem services estimated in Chapter 2 and 3, and the third part summarizes the biological and economic parameter assumptions used for empirical analysis.

5.1.1 Spread dynamics

The discrete-time and spatially explicit Monte-Carlo simulation approach models the floatplane-related spread of elodea using a finite metapopulation consisting of seven habitat patches—regions (Figure 5.1) (Facon and David 2006; Levins 1969). The model is then expanded to include how management of invaded floatplane hubs changes colonization rates across regions.

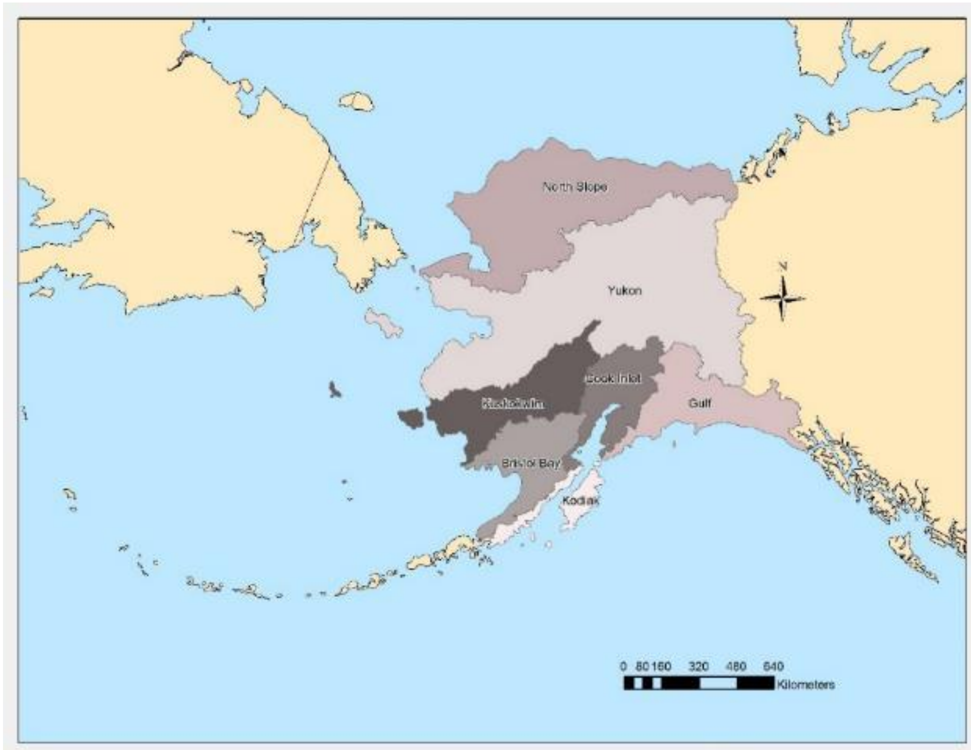

Figure 5.1 Study regions used to aggregate floatplane flight patterns. The southern five regions closely encompass watersheds supporting commercial salmon fisheries.

Following the traditional metapopulation approach, patches are either empty (state β = 0, where elodea is absent or too rare to be detected) or occupied (state β = 1, elodea detected). Patch occupancy is determined by colonization and extinction rates. Since colonization success is directly linked to propagule pressure (Schreiber and Lloyd-Smith 2009), floatplane flight frequencies in Table 5.2 are used to proxy region-specific colonization rates, c^i. The region-specific colonization rates equal the proportion of flights to region i that originate from elodea-invaded floatplane hubs (Table 5.2), over the total number of annual flights to destinations within region i, or in mathematical terms $c^i = \frac{\sum_{i=1}^{7} v^i{}_e}{\sum_{i=1}^{7} v^i}$, for regions $i = 1, ..., 7$. If a region is occupied in year t, there is a probability, e_{t+1}, that elodea will go extinct in year $t+1$. If the patch is unoccupied, the probability it is colonized in $t+1$ is $c^i{}_{t+1} P_t$ where P_t is the proportion of patches occupied in year t.

During time interval $[t, t+1]$, each patch can show one of the following four transitions in *β*: remaining empty $(0 \rightarrow 0)$, newly occupied $(0 \rightarrow 1)$, newly extinct $(1 \rightarrow 0)$, and remaining occupied $(1 \rightarrow 1)$. For the first two transitions mentioned,

$$\beta^{i}{}_{t+1} = \begin{cases} 1 & \text{if } u^{i}{}_{t+1} < c^{i}{}_{t+1} P_t \\ 0 & \text{otherwise} \end{cases},$$

where *u* is a random variable described by a uniform distribution bounded by zero and one. If the patch was occupied the previous year, then

$$\beta^{i}{}_{t+1} = \begin{cases} 0 & \text{if } u^{i}{}_{t+1} < e_{t+1} \\ 1 & \text{otherwise} \end{cases}.$$

5.1.1.1 Incorporating management

In order to incorporate the effects of management actions (including the option of no action), the vector, M, accounts for the management status across patches, thus $\eta = f(M, \beta)$. The patch now can take one of three states: never been occupied (η = 0, elodea currently not known to ever have occurred), occupied (η = 1, elodea detected), or recovering (state η = -1, elodea managed or naturally extinct). Note, once a patch is occupied by elodea once, it is either in recovery or is re-occupied, thereafter. Also, once management occurs, it is also to occur in subsequent years.

Colonization rates depend on which of the seven floatplane bases in Table 4.2 is invaded and which of the invaded ones were managed in the previous year. The model assumes a two year lag between invasion and detection in floatplane hubs, consistent with research suggesting peak biomass can occur within four years (Mjelde et al. 2012; Heikkinen et al. 2009; Simpson 1984). Management vector, M contains each management action in each patch i, as follows $m^{i}{}_{t} = \begin{cases} 1 & \text{if action } \leq t \text{ and } \beta_t = 1 \text{ for all 4 years prior} \\ 0 & \text{otherwise} \end{cases}$. The colonization rate in year t for patch i

is now dependent whether any management occurred in the previous year, thus $c^{i}_{t+1} = f(v^{i}_{e}(\mathrm{M}_t), v^{i})$. The transitions for $\eta = f(\mathrm{M}, \beta)$ during time interval $[t, t+1]$ are as follows.

If the patch remains unoccupied, thus *β* remains $(0 \rightarrow 0)$, and the patch was not colonized prior to t, *η* remains $(0 \rightarrow 0)$, or *η* switches from $(-1 \rightarrow -1)$ if the patch was recovering. If the patch is newly colonized, thus *β* switches from $(0 \rightarrow 1)$, and management occurs in $t+1$, then *η* switches from $(0 \rightarrow -1)$, or *η* switches from $(-1 \rightarrow -1)$ if the patch was recovering. If the patch is newly extinct, thus *β* switches from $(1 \rightarrow 0)$, and the patch was occupied and unmanaged in t, *η* switches from $(1 \rightarrow -1)$, or *η* remains $(-1 \rightarrow -1)$ if the patch was recovering. If the patch remains occupied, thus *β* remains $(1 \rightarrow 1)$, and there is no management, *η* remains $(1 \rightarrow 1)$, or *η* switches from $(1 \rightarrow -1)$ if there is management in $t+1$. Mathematically, the above is as follows:

$$\eta^{i}_{t+1} = \begin{cases} 1 & \text{if } \mathrm{m}^{i}_{t+1} = 0 \text{ and } \beta^{i}_{t+1} = 1 \\ -1 & \text{if } \mathrm{m}^{i}_{t+1} = 1 \text{ and } \beta^{i}_{t+1} = 1 \\ & \quad \text{or } \beta^{i}_{t+1} = 0 \text{ and } \beta^{i}_{t} = 0 \text{ and } \eta^{i}_{t} = -1 \\ & \quad \text{or } \beta^{i}_{t+1} = 0 \text{ and } \beta^{i}_{t} = 1 \\ 0 & \text{otherwise} \end{cases}.$$

5.1.2 Integrating ecosystem services

In order to account for changes to ecosystem services associated with elodea-invaded regions, the functional responses to elodea are as follows. For floatplanes, the number of landing-spots invaded within a patch is described for the next period as $l_{t+1} = f(l_1, l_t, L, \eta_{t+1})$, where l_1 is the number of destinations currently invaded and L is the number of floatplane landing-spots in each patch (Table 4.2). This means, the model accounts for the inter-regional dispersal of elodea among floatplane destinations within a region (Deines, Chen, and Landis 2005).[45] For simplicity, region-specific notation is omitted. The number of landing-spots that can be invaded is bound by $1 \le l_{t+1} \le L$ if $\eta_{t+1} = 1$ and $l_{t+1} = 0$ if $\eta_{t+1} = -1$. This functional relationship allows for the number of invaded floatplane destinations within a region to increase when one of the destinations in the region is invaded. Once a region is managed, all destinations return to their uninvaded status. For commercial fisheries, harvest in the next period is described as $h_{t+1} = f(h_0, h_t, K, \theta, \eta_{t+1})$ where h_0 is the current harvest level in lbs (without elodea), K is the maximum observed harvest level in the past ten years, and θ is the annual average growth rate for sockeye salmon biomass in elodea-invaded habitat (Chapters 2 and 3). Harvest is bound by $0 \le h \le h_0$, later referred to as the harvest constraint. Figure 5.2 illustrate these dynamics for Cook Inlet and three different functional relationships, assuming management occurs in year 50.

In order to account for uncertainty in population dynamics following an ecosystem perturbation, three types of of growth functions are tested (Case 2000). The first type assumes fixed annual impact such that $l_{t+1} = l_t + Lc_{t+1}$ and $h_{t+1} = h_t + \eta\theta h_0$. The second type considers exponential growth specified as $l_{t+1} = l_t\left(1 + c_{t+1}\right)$ and $h_{t+1} = h_t\left(1 + \eta\theta\right)$. The third type describes a logistic growth relationship as $l_{t+1} = l_t\left(1 + c_{t+1}\left(1 - l_t / L\right)\right)$ and $h_{t+1} = h_t\left(1 + \eta\theta\left(1 - h_t / K\right)\right)$ (Figure 5.2).

Welfare changes accruing to recreational floatplane pilots are estimated in Chapter 3 using a multinomial probit model and maximum likelihood optimization (Hausman and Wise 1978). When a floatplane destination becomes invaded, the marginal user loss per flight accruing to recreational floatplane pilots is w. The annual floatplane user loss is equal to

$$\Delta CSP_{t+1}{}^{i} = l^{i}{}_{t+1} \frac{\sum_{i=1}^{7} v_r{}^{i}}{L^{i}} w \tag{5.1}$$

[45] The process of an invader establishing a population in a remote patch before colonizing the remaining landscape is also known as the beachhead effect (Deines, Chen, and Landis 2005).

where the fraction is equal to the annual average recreational flights per destination with v_r being recreational flights. The management cost associated with eradicating elodea in year *t* in each region is equal to $C_t^i = s\left(l^i{}_t \cdot a^i + a^{ihub}\right)$ and assumes once each waterbody is managed or goes extinct once, it remains uninvaded unless re-colonized. In other words, the model assumes local eradication is successful. The region-specific average landing-spot size in surface acres (Chapter 3) is a^i and a^{ihub} is the size of each floatplane base. The per-acre cost of herbicide is *s*.

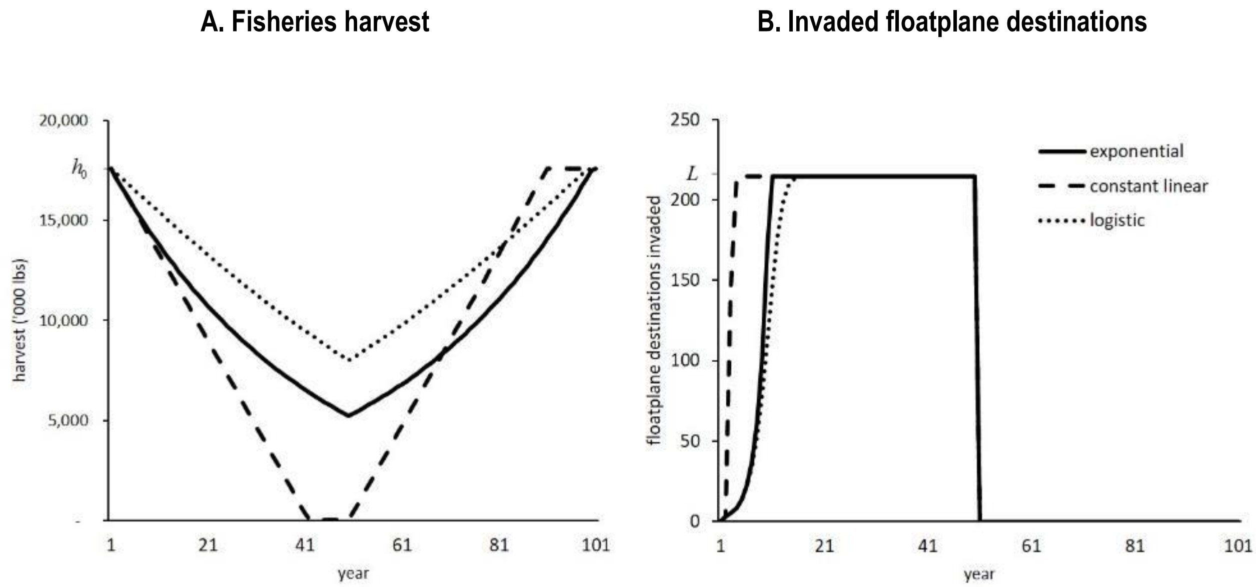

Figure 5.2 Dynamics of harvest and invaded floatplane destinations within a region under different growth type assumptions. Management occurs in year 50.

Impacts to commercial fisheries are estimated in Chapter 3. The forgone net benefits to consumers resulting from a decrease (increase) in annual harvest and a consequential increase (decrease) in the price per lbs. are estimated. The model assumes a linear and downward sloping demand function, thus the own-price elasticity of demand can be used to explain how responsive demand for sockeye products is given changing in its prices (Freeman 2003). In mathematical terms, annual damages per region are equal to

$$\Delta CSF_{t+1}{}^i = \frac{\gamma}{2}\left[\left(h^i{}_t - f(h^i{}_0, h^i{}_t, K^i, \theta, \eta^i{}_{t+1})\right)\left(\left(\frac{h^i{}_t}{\varepsilon}p^i{}_t\right) + \frac{f(h^i{}_0, h^i{}_t, K^i, \theta, \eta^i{}_{t+1})}{h^i{}_t \varepsilon}\right) - p^i{}_t h^i{}_t\right], \tag{5.2}$$

where γ is processing yield, p is the real (inflation-adjusted) per lbs wholesale price for sockeye salmon in 2015 USD received by Alaska primary processors, and ε is the own-price elasticity of demand. Prices are weighted by sockeye product ratios commonly observed in the Alaska processing sector (Chapter 3).

The potential damages are expressed in net present value (NPV) terms thus illustrating the loss in natural capital. The combined discounted over a 100-year time horizon and summed across the seven regions, can be expressed as:

$$NPV = \sum_{t=0}^{100}\sum_{i=1}^{7}\left(\Delta CSF_t^i + \Delta CSP_t^i + C_t^i\right)\left(1+d\right)^{-t}, \tag{5.3}$$

where *d* is the social discount rate. The constant annualized loss in ecosystem services is estimated as follows,

$$NPV_{annual} = NPV\frac{d}{1-\left(1+d\right)^{-100}} \tag{5.4}$$

5.1.3 Management alternatives

Equation (5.3) and (5.4) are estimated for two management alternatives, no action and strategic action. In addition, each alternative has two cases, a base-case and a worst-case that only differ in the way the model is initialized at t_1. In the base-case, the patches that are occupied at t_1 reflect the currently known elodea-invaded floatplane hubs of Lake Hood (Cook Inlet region) and Eyak Lake (Gulf region). In a hypothetical worst-case, all regional floatplane hubs are initially occupied, thus $\beta_1^i = 1$ for all *i*. Note, in any case, all patches are considered empty in year t_0 thus $\beta_0 = \eta_0 = 0$ for all i, and I_1 equals the number of currently invaded floatplane destinations in that region.

For the strategic action alternative, the manager's optimization problem becomes $\min NPV$ s.t. $M(t)$, where management *M* is a function of time when action is taken, assuming each region is managed one year at a time. The optimization uses the OptQuest algorithm to find the optimal order in which management should occur across the seven regions to minimize Equation (5.3) (Glover, Kelly, and Laguna 1996; Palisade 2016).

5.1.4 Parameter assumptions

Biological and economic parameter values were derived from several sources (Table 4.3) as well as Chapters 2 and 3. Rather than discussing each parameter, it is worth pointing out a few important ones. Due to the recent discovery of elodea in Alaska, disappearing elodea populations have not yet been observed, preventing the estimation of a local extinction rate, e. However, data from Norway where elodea is growing in similar climatic conditions serves as a proxy. In 2012, among the 47 elodea-invaded lakes in Norway, *E. canadensis* disappeared from three lakes for a mean extinction probability of 0.0638 (Mjelde et al. 2012). Similar crashes of invasive elodea have been observed in Germany where *E. canadensis* used to be a major weed in the past but is now a rare species that disappeared from many waters (Hussner 2017 personal communication). Using the Norwegian data, a beta distribution is used to account for uncertainty. This distribution was truncated at a maximum threshold of 0.5 to lower the probability of Alaska-wide elodea extinction events. Probabilistic predictions in ecological models often need to be scaled to be consistent with real observations or realistic outcomes. Due to lack of data for model calibration, this threshold was arbitrarily chosen (Guisan and Zimmermann 2000). Since the model applies a region-wide extinction rate, the truncation allowed for this region-wide extinction to be less likely compared to for example localized extinction events reflected in the data (Mjelde et al. 2012).

Due to lack of empirical evidence about the ecological effects of elodea on salmonid reproduction in non-regulated freshwater habitat (Merz et al. 2008; Carey et al. 2016), the analysis of potential damages to commercial fisheries relies on structured expert judgment (Cooke 1991). First, a scenario-based elicitation related to elodea's potential ecological effects on salmonid persistence in elodea-invaded habitat was conducted. It quantified how expert opinion on persistent salmon populations is sensitive to varying habitat and invasion characteristics (Chapter 1).[46] A follow-up interval judgment then elicited annual average growth rates, θ, expected for

[46] Environmental characteristics included location of elodea within the salmon system, description of the salmon system, dissolved oxygen levels, predation, prey abundance, and other factors.

sockeye salmon in elodea-invaded habitat.[47] Individual expert opinion was combined applying equal weights to result in a joint probability distribution of growth rates (Chapter 2) (Morgan 2014). This joint distribution represents the uncertainty in elodea's overall effect on sockeye salmon. Additionally, the joint probability distribution also reflects varying opinions on elodea's overall effects on salmon, with a 0.265 probability of observing positive growth in elodea-invaded habitat. This result was consistent with the first scenario-based elicitation which found that the mean probability of salmon persisting in invaded habitat was 0.21 (Chapter 1).

While there are reliable market data related to commercial fisheries harvest and prices, there is a lack of recent studies explaining the responsiveness of consumer demand to changes in the price for salmon. Prices for wild Alaska salmon have been depressed in the 1990s with the onset of salmon farming. Yet over the past decade, prices recovered due to marketing efforts aimed at wild and sustainably caught Alaska salmon, in addition to disease outbreaks in salmon farms elsewhere (Knapp, Roheim, and Anderson 2007). In 2014, wild salmon comprised about 30% of global salmon production by volume, with 65% from Alaska. While, very close substitutes to wild sockeye salmon like coho, pink, or chum would support higher elasticities more inelastic demand is supported by brand loyalty to a wild and sustainably harvested product. For these reasons, the analysis uses the minimum (-12.78) and maximum (-1.472) own-price elasticity of demand found in the literature to bound a uniform distribution to account for uncertainty in ε (Table 4.4).

Management costs of aquatic plants vary by many different factors most significantly by the type of removal method, species, abundance, and management goal. Additional factors are site-specific such as the extent of invasion (partial vs. full lake treatment)[48], water depth, water volume, remoteness, water flow and related herbicide dissipation, and herbicide formulation (pellet vs. liquid) (Schardt 2014 personal communication). Compared to herbicides, mechanical removal is much more costly and less effective for submersed aquatic plants such as elodea (Hussner et al. 2017). The per-acre cost of mechanical removal ranges between $12,000 and $20,000 in 2015 USD and due to its poor success record for elodea was not considered in this

[47] The annual average growth rate was referred to as salmon production over many life cycles manifesting itself as a long-term "trend in abundance" (McElhany et al. 2000).

[48] Partial lake treatments often require the use of a contact herbicide such as Diquat to prevent localized elodea populations from spreading throughout a lake, adding to the per-acre cost (J. M. Morton et al. 2014).

analysis (Johnson 2013; Lane 2014, Schardt 2014 personal communication). Using historical expenses for treating *Hydrilla verticillata* (hydrilla) between 1980 and 2002 a Weibull distribution was fitted to the historical inflation adjusted per-acre cost. Since hydrilla is primarily treated with Fluridone, at similar concentrations to elodea, the mean per-acre cost of $861 in 2015 USD is comparable for the treatment of elodea.[49] The observed costs are comparable to treatment costs in Alaska (J. M. Morton et al. 2014).

Finally, the discount rate, *d*, is another key uncertainty for which a triangular distribution was used assumed a range between 1% and 6% with a peak of 3% (Table 5.2). This assumption is consistent with the real 30-year discount rate recommended by OMB and similar analysis of invasive species risk recently conducted (Rothlisberger et al. 2012; OMB 2016).

Table 5.1 Floatplane landing site characteristics

Regional floatplane hub	Size (acres), a^{ihub}	Region	Median size of destination (acres), a^i	Annual avg. flights/ destination	Number of destinations, L^i	Base-case a) l_1^i
Shannon's Pond, Dillingham	31	Bristol Bay	3666	216	74	0
Lake Hood, Anchorage	270	Cook Inlet	185	180	215	3
Hangar Lake, Bethel	137	Kuskokwim	916	42	28	0
Eyak Lake, Cordova	2495	Gulf	381	49	71	3
Lilly Lake, Kodiak	15	Kodiak	307	145	41	0
Float Pond, Bettles	155	North Slope	828	24	93	0
Float Pond, Fairbanks	136	Yukon	336	205	205	0

a) Count of currently known elodea invaded waterbodies that are floatplane destinations in each region. Count does not include floatplane hub.

[49] This timeframe was used rather than including 2003-2013. Between 2003 and 2014, two massive hurricanes added to cost as well as a change of management occurred (Shuler 2015 personal communication).

Table 5.2 Model parameters for empirical analysis

Parameter	Units	Region-specific	Mean or values	Distribution	Source
Spatial dynamics					
Colonization rate, *c*	decimal	yes	Model	n/a	Model-determined
Extinction rate, *e*	decimal	no	0.0638	Beta (0.002, 0.02 97)[a)]	Mjelde et al. 2012
Proportion of colonized patches, *P*	decimal	n/a	Model	n/a	Model-determined
Patch state before management, β	binary	yes	0 or 1	n/a	Model-determined
Vector of management actions, *M*	binary	yes	0 or 1	n/a	Model-determined
Patch state after management, η	ternary	yes	0, 1, or -1	n/a	Model-determined
Random variable, *u*	decimal	yes	(0, 1)	Uni (0, 1)	This study
Floatplanes					
Floatplane destinations, L^i	sites	yes	Table 5.1	n/a	Chapter 3
Initial invaded destination count, l_1^i	sites	yes	Table 5.1	n/a	Chapter 3
Mean surface size of floatplane destinations, a^i	acres	yes	Table 5.1	n/a	Table 4.3
User loss per flight, *w*	2015 USD	no	$185/flight	Normal (185, 13.78)	Chapter 3
Surface size floatplane hub, a^{i-base}	acres	yes	Table 5.1	n/a	Table 4.4
Commercial fisheries					
Annual average sockeye growth rate, θ	decimal	no	-0.024	Normal (-0.024, 0.039)	Chapter 2
Initial harvest level, h_0	lbs	yes	Chapter 2	Lognormal	ADFG 2016a
Wholesale price for sockeye products [b)], *p*	2015 USD	yes	Chapter 2	Lognormal	ADFG 2016b
Own-price elasticity of demand, ε	decimal	no	Model	Uni (-12.78, -1.472)	Wang 1976; DeVoretz 1982
Ecological limit of sockeye harvest, *K*	lbs	yes	Chapter 2	n/a	Chapter 2
Processing yield, γ	decimal	yes	Chapter 2	n/a	Knapp, Roheim, and Anderson 2007
Management					
Herbicide cost, *s*	2015 USD	no	$861/acre	Weibull (9.71, 907)	Schardt 2014; J. M. Morton et al. 2014; CH2MHILL 2005
Discount rate, *d*	decimal	no	0.03	Tri (0.01, 0.03, 0.06)	Rothlisberger et al. 2012; OMB 2016

a) Truncated at 0.5. b) Weighted by the region-specific product amounts for frozen, canned, fresh, and other (Chapter 3).

5.2 Results

Results are presented for the no action alternative and strategic alternative, each assuming base-case and worst-case initial colonization. Even though the worst-case is hypothetical, it provides an upper bound to damages given uncertainty in the true state of currently unknown invasions. The sensitivity analysis tests the model's robustness for varying assumptions related to parameters, growth types, and harvest constraints. Simulation estimates were monitored for convergence every 100 iterations with a convergence tolerance of 3% and a confidence level of 95% surrounding the median NPV. The model converges using less than 10,000 iterations.

5.2.1 No action alternative

5.2.1.1 Base-case

Given the spread dynamics outlined above, the parameterized metapopulation model predicts invasions across the seven regions and 100-year time horizon. The probability of invasion varies among regions dependent on the floatplane pathway and the extinction probability. Figure 5.3 illustrates this for the no action base-case. Due to the currently existent elodea populations the model predicts the highest invasion probabilities for the Cook Inlet and Gulf regions throughout the time horizon. The Kuskokwim and Bristol Bay regions currently have a higher than one in two chance of being invaded. The model predicts 100% probability of these two regions being invaded 50 years from now, should action not occur. The Yukon, North Slope, and Kodiak regions currently show low probabilities of being invaded through floatplanes. This simulation outcome, however, does not account for other vectors that are currently at play particularly in the Yukon region. Unmanaged invasions in the Yukon region have not made it into waterbodies known to be used by floatplanes. These unmanaged invasions continue to be source locations for elodea and subject to other vectors. For example, most likely river current transported elodea to new locations in the Yukon region (Friedman 2015). In contrast, floatplane operations in the Kodiak region are often saltwater based, providing a natural risk buffer to the spread of elodea.

Applying Equation (5.3), the current value of potentially lost natural capital amounts to a median—that is the most probable—NPV of $1,382.5 million in 2015 USD (90% CI: 126.0; 9,419.4 million) for all regions combined. The associated mean NPV amounts to $4,449 million, indicating the damage distribution is skewed. The constant annual loss in ecosystem services associated with this value is estimated using Equation (5.4) and amounts to a median annualized NPV of $50.2 million (90% CI: $5.5; $263.9) (Table 5.3). Simulation results clearly show that the greatest risk of invasion is in the not yet invaded Bristol Bay region. Although the range of uncertainty is large, the median is positive and largest among regions, followed by estimates for the Cook Inlet region. The Cook Inlet region differs from Bristol Bay as a much larger portion of damages (26%) relates to recreational floatplane pilots not being able to access their elodea-invaded landing zones (Table 5.6). The Gulf region shows the third highest potential risk from elodea-invasions. Comparing the total NPV estimates for fisheries and floatplane pilots with results in Chapter 2 and 3 shows that a spatially-explicit approach to risk analysis significantly reduces the magnitude and variance in simulation outcomes.

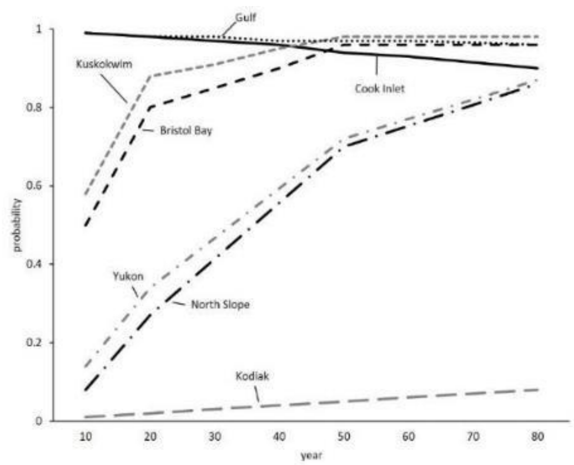

Figure 5.3 Probability of invasion by region, no action base-case

Table 5.3 Mean damages to fisheries and recreational pilots (NPV million of 2015 USD), no action base-case

Region	Fisheries	%	Pilots	%	Total
Bristol Bay	1,562.2	99%	1.0	<1%	1,563.3
Cook Inlet	474.9	74%	167.9	26%	642.8
Kuskokwim	3.4	97%	0.1	3%	3.5
Gulf	342.7	96%	14.2	4%	356.9
Kodiak	60.1	99%	0.0	<1%	60.1
North Slope	0	0%	0.1	100%	0.1
Yukon	0	0%	0.5	100%	0.5
Total	2,443.3		183.3		2,626.6

Table 5.4 Potential annual loss in ecosystem services and potential loss in natural capital (million of 2015 USD), no action base-case

Region	Ecosystem Services (annualized NPV)			Natural Capital (NPV 100-year period)		
	Median	5%	95%	Median	5%	95%
Bristol Bay	22.2	0	176.7	620.7	0	6,288.5
Cook Inlet	14.7	4.9	56.3	412.4	108.6	1,925.1
Kuskokwim	0.0	0	0.4	1.3	0	13.9
Gulf	7.9	0.4	33.9	219.1	9.7	1,191.8
Kodiak	0.4	0	8.2	10.6	0	280.4
North Slope	< 0.1	0	< 0.1	0	0	0.0
Yukon	< 0.1	0	< 0.1	0	0	0.7
Total[a)]	50.2	5.5	263.9	1,382.5 [b)]	126.9	9,419.4

a) Note, totals do not sum over individual values per region since the results depict distributions rather than discrete values. b) Mean NPV is $4,449 million.

5.2.1.2 Worst-case

The floatplane related damages are much smaller in the above shown base-case simply because the simulation starts out with the two currently invaded floatplane hubs being invaded. Considering the worst-case—that all regional floatplane hubs are invaded—as the model is initialized, then damages to recreational floatplane pilots are much more pronounced. Table 5.7 illustrates that under such a hypothetical state, all regions would face significant risk from elodea-invasions particularly the Bristol Bay and the Yukon regions. In these regions, potential recreational user loss would increase to a median NPV of $72.1 and $53.0 million in 2015 USD respectively. Also, the Kodiak region would face significant risk in contrast to relatively low invasion probabilities (Figure 5.3). Overall, the more prominent recreational user loss in the worst-case, would increase the median NPV of combined damages by 0.9 billion to a median of $2.2 billion in 2015 USD (95% CI: $195 million, $12.7 billion). This result underlines the importance of integrating spread dynamics and region-specific economic values in bioeconomic risk analysis of biological invasions.

Table 5.5 Median damages to recreational pilots (NPV million of 2015 USD), no action worst-case

Region	NPV		
	Median	5%	95%
Bristol Bay	72.1	42.2	126.9
Cook Inlet	160.0	85.5	292.8
Kuskokwim	5.9	3.7	9.9
Gulf	13.6	7.4	25.2
Kodiak	13.0	5.3	28.2
North Slope	9.8	5.7	17.5
Yukon	53.0	31.3	93.1
Total[a)]	263.4	146.6	478.6

5.2.2 Strategic action alternative

Under strategic action management, the manager optimizes to minimize long-term damages by deciding when and where to eradicate elodea invasions, given agencies have only enough resources to treat one region one year at a time. While the latter analysis is somewhat hypothetical, it illustrates the importance of distinguishing between region-specific risk in statewide management decision making and aids prioritization of action. Base-case and worst-case model initialization is considered.

5.2.2.1 When to take action

Figure 5.4 A suggests that the optimal time to take action is now (t = 1), where the median NPV is minimized at at $1.9 million in 2015 USD. There is 5% probability that current damages could already exceed $1.3 billion and 5% probability current damages could be less than $0.2 million. The median NPV and its 90% uncertainty range is increasing the longer management action is delayed, underlining the fact that chances are increasing to realize much greater damages than shown by the median. In other words, the probability of extreme damages is increasing. Additionally, even though, the model predicts a larger range of possible damages in the future, the probability of observing no damages in the future is zero.

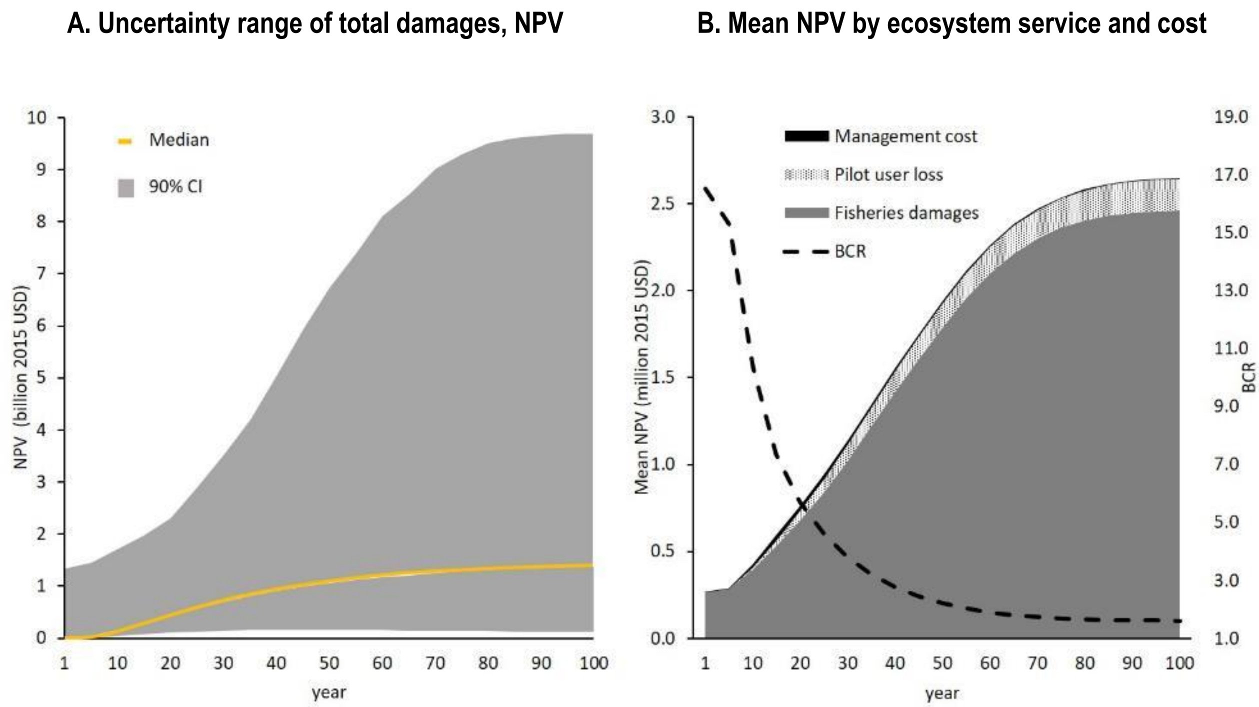

Figure 5.4 A. NPV of potential damages over time B. Mean NPV for fisheries damages, pilot user loss, and management cost dependent on year action occurs. Note, the differences in NPV scale, where A. shows the median and uncertainty range and B. shows the mean. Due to the skewed damage distribution, the mean is used to establish the benefit-cost ratio.

Figure 5.4 B shows the estimated mean management cost in relation to pilot user loss and fisheries damages, given management occurs in year, *t*. The upfront mean management cost of $3.5 million to treat currently invaded waterbodies associated with floatplane operations in the Cook Inlet and Gulf regions, is insignificant compared to the currently observed mean damages of $265 million. Would such management action occur immediately, the benefits would amount to the mean avoided damages, in specific the mean NPV under the no action base case minus the already accrued mean damages. Furthermore, the cost would amount to the current management costs plus the currently accrued damages. With immediate action, the benefits would outweigh the costs by a factor of nearly 17 as shown in Figure 5.4 B. Immediate action is crucial because the benefit-cost-ratio (BCR) is quickly decreasing within the next ten years, emphasizing that the most effective window for action is closing rapidly. Twenty-five years from now, the mean benefits of taking action will only outweigh mean costs by a factor of four, yet the uncertainties surrounding this estimate are much greater due to the wider range of possible damages 25 years from now.

5.2.2.2 *Where to take action*

An additional question regarding strategic action is where to manage first, assuming management agencies are only able to manage one region at a time. Considering the base-case, optimal management would first target Lake Hood and invaded waterbodies in the Cook Inlet region, and then Eyak Lake and invaded waterbodies in the Gulf region. This result is consistent with what management agencies actually decided to do when they treated Lake Hood in 201 (DNR 2015). However, more action is required. Figure 5.5 illustrates the cost of delaying management of Eyak Lake and invaded waterbodies in the Gulf region, assuming management in Cook Inlet precedes action in the Gulf region. The median statewide damages associated with not taking action in the Gulf region are forecast to amount to a median damage estimate that exceeds $7 million in 2015 USD four years from now.[50] The benefits (avoided damages) by taking action now amount to a median NPV of $1,380.6 million.

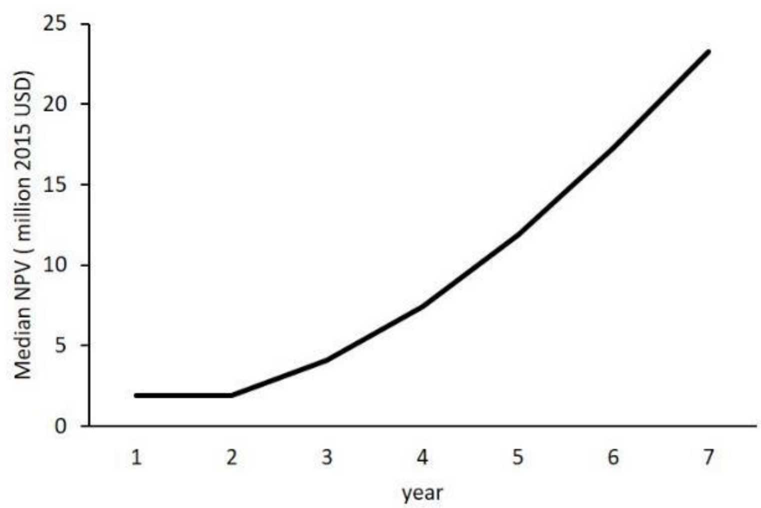

Figure 5.5 Median NPV damages dependent on when Eyak Lake is managed.

Under the worst-case, assuming all regional floatplane hubs would be invaded, the model was used to establish an optimal management schedule. Such an exercise illustrates which

[50] Management agencies have estimated the cost of managing elodea-invaded waterbodies in the Copper River Delta to amount to $7 million.

regions carry the largest risk associated with localized elodea invasions. Table 5.8 shows the optimal treatment schedule calculated by the model in the third column, and schedules given other forms of prioritization. For the optimal management schedule, the highest priority for eradication would be given to Dillingham's Shannon Pond and the Bristol Bay region. This result is not surprising considering the largest economic risk the Bristol Bay region carries for elodea invasions. Interesting to note, under this worst-case, the priority order between Lake Hood and Eyak Lake changes, placing Eyak Lake ahead of Lake Hood. This result outlines once more the importance of spatially-explicit models that cover spread dynamics but also the need to account for the influence of management.

Table 5.6 Evaluation of management strategies, hubs treated one year at a time, strategic action worst-cases

Floatplane hub	Region	Treatment schedule (year)		
		Optimal as determined by model	Total flights	Flights to other regions
Shannon's Pond, Dillingham	Bristol Bay	1	3	4
Lake Hood, Anchorage	Cook Inlet	3	1	1
Hangar Lake, Bethel	Kuskokwim	6	7	5
Eyak Lake, Cordova	Gulf	2	6	7
Lilly Lake, Kodiak	Kodiak	4	4	6
Float Pond, Bettles	North Slope	5	5	3
Float Pond, Fairbanks	Yukon	7	2	2
Median NPV (million 2015 USD)		14.7	33.5	52.5

Given management would not have information on the optimal treatment schedule in absence of the presented study, other forms of prioritization could be used. For example, using the total number of flights originating from each hub, the median damages would more than double from a median NPV of \$14.7 million to \$33.5 million. Damages would triple if the prioritization is based on the number of flights to other regions (Table 5.8). This results shows the utility to management of quantitative bioeconomic risk analysis for biological invasions.

5.2.3 Sensitivity analysis

The sensitivity analysis is comprised of three parts, first considering the sensitivity of damage estimates to parameter assumptions, second describing how these results deviate by assuming different types of growth, and finally investigating the impact of relaxing the harvest constraint.

5.2.3.1 Parameter assumptions

Given the above results, it is no surprise that fisheries-related parameters are most influential on the damage estimate. For the no action base-case, the annual average growth rate for sockeye salmon in elodea-invaded habitat, θ, contributes about a third of the variance in the damage estimates, followed by the discount rate (9%), d, and Bristol Bay wholesale price for frozen sockeye products (8%) (Table 4.9). The Bristol Bay harvest assumption, h_0, and the price assumptions for its canned product are also among the most influential factors in the model but have lower effects compared to the previously mentioned parameters.

The contribution of the annual average growth rate to variance is not surprising considering the high range of uncertainty in expert-derived sockeye growth rates. There is a strong negative correlation between the growth rate and NPV (Table 5.9, Figure 5.6 B). The negative growth rates have a larger impact on the mean NPV compared to positive growth rates. For example, the lowest growth rate of -0.12 increases the mean NPV by $7 billion compared to the highest growth rate of 0.07 which only decreases mean NPV by $200 million. The observed sensitivity of the growth rate is mainly driven by the harvest constraint discussed below which limits the extend to which habitat changes can lead to an increase in harvest beyond historically observed harvest levels. This is particularly the case when the growth rate is positive and the system is invaded, or when the growth rate is negative and the system is in recovery. In the latter case, the system would recover beyond historically observed harvest levels, h_0, as the harvest constraint is relaxed.

As expected, simulation outcomes also vary with the assumed discount rate, where larger discount rates lead to future damages being discounted more than damages that occur sooner. Thus larger discount rates lower NPV more so than smaller discount rates, resulting in negative correlation (Figure 5.6 B). With Bristol Bay being the largest sockeye salmon fishery in Alaska and frozen products being its main line of business, the contribution to variance of Bristol Bay prices is not surprising. A price assumption of $18.59/lbs in 2015 USD results in an increase of the mean NPV by $4.3 billion, whereas a price of $0.82/lbs reduces NPV by $1.6 billion (Table 4.9).

Table 5.7 Sensitivity of damage estimates to parameter assumptions with the largest influence on mean total NPV, no action base case

	% Contribution to variance	Correlation (Spearman Rank)	Change in mean NPV (billion USD) a) Lowest input assumption	Highest input assumption
Annual average sockeye growth rate, θ	-35.6%	-0.85	7.0	-0.2
Discount rate, d	-9.4%	-0.35	5.2	-0.9
Price for frozen product in Bristol Bay	7.8%	0.16	-1.6	4.7
Historical Bristol Bay harvest, h_0	1.26%	-0.07	3.1	-1.7
Price for canned product in Bristol Bay	1.19%	0.06	-2.3	3.4

a) Changes in the mean NPV are calculated holding all other parameters constant at their mean levels.

Due to its contribution to variance in the damage estimates, the growth rate assumption warrants further analysis comparing sensitivity in the no action base case with sensitivity in the strategic action case (Figure 5.6). The reason for a closer look is due to the fact that the SEJ-elicited probability distribution for θ shows a 26.5% chance of elodea being a good to salmon rather than a bad. Thus, for positive growth rates, management can have unintended consequences of reducing the benefits of elodea to salmon, thus fisheries realize forgone benefits.

In the no action base-case, positive growth rates have little effect on NPV as shown by the solid line for θ greater than the 75^{th} percentile (Figure 5.6 B). This result is largely due to the 100 year unmanaged time horizon where habitat can either be elodea-invaded or where there is a low probability elodea goes extinct. Thus, the negative growth rates have more influence over the 100-years. However, when management action occurs in year 1, there are 99 years remaining in which the habitat is in recovery or uninvaded. Negative growth rates have no effect on damages, whereas positive growth rates have a much larger effect that increases damages (forgone benefits) more steeply than price or the discount rate. Note, however, that the magnitude of the latter described effect is much smaller than the magnitude of the effect in the no action case.

5.2.3.2 Growth assumptions

Simulation results are also sensitive to assuming different types of growth underlying elodea's effects on sockeye salmon and the inter-regional dispersal of elodea among floatplane destinations within a region, both illustrated by Figure 5.2. If a region is invaded in t = 1 and remains unmanaged, the linear growth assumption results in a sharp and linear drop in harvest (rise in invaded destinations for floatplane pilots), whereas logistic and exponential growth both are non-linear with more and less moderated declines (rise in invaded destinations) respectively (Figure 4.2). Assuming linear growth, estimated damages are more than twice the magnitude compared to the logistic growth assumption (no action base case). Similarly, exponential growth assumptions are by three thirds higher compared to damages assuming logistic growth (Table 5.10). For damages to fisheries, the linear and exponential growth assumptions result in similar estimates, with estimates under the linear assumption being more uncertain. For damages to floatplane pilots, the exponential and logistic assumptions result in similar estimates with similar uncertainty. Table 4.10 illustrates that the logistic growth assumption used in the base-case provides a lower bound for the potential damages.

A. No action **B. Strategic action**

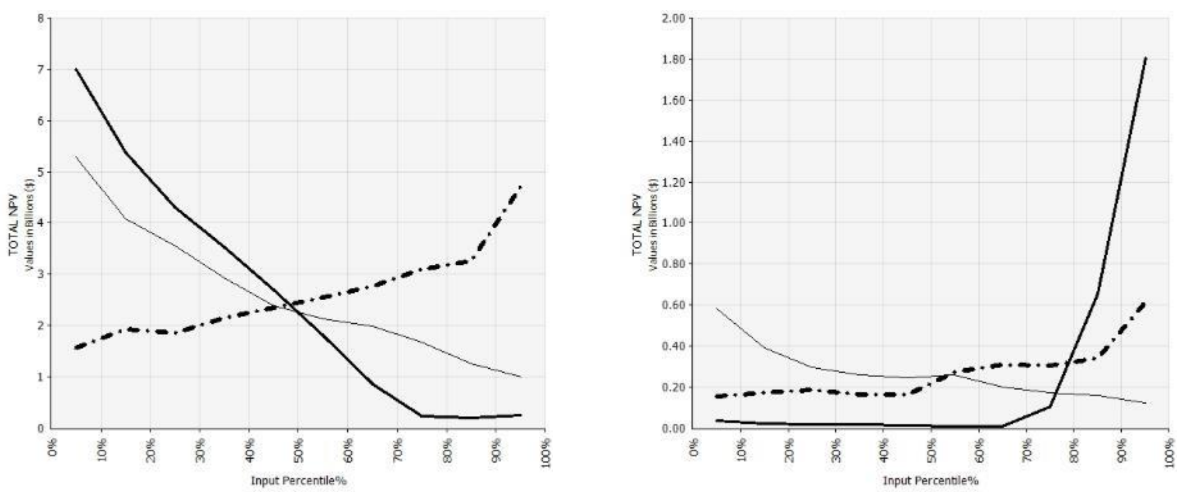

Figure 5.6 Change in mean total NPV dependent on percentile changes for annual average growth rate for sockeye salmon (solid line), *θ*, discount rate (thin line), *d,* and prices for frozen product (dotted line). Note, the differences in NPV scale, thus the steepness of curves cannot be compared across (A) and (B).

Table 5.8 Sensitivity of NPV to growth assumptions, millions 2015 USD, no action

Growth type	Total NPV		Fisheries NPV		Pilots NPV	
	Mean (%)	SD	Mean (%)	SD	Mean (%)	SD
Constant linear	5,792 (121%)	6,307	4,495 (84%)	6,265	297 (62%)	101
Exponential	4,652 (77%)	5,280	4,462 (83%)	5,253	190 (4%)	72
Logistic	2,627	3,371	2,443	3,345	183	71

Note, %-change compared to logistic.

5.2.3.3 Harvest constraint

As discussed above, the harvest constraint limits the damage-reducing effect of positive values in θ. For logistic growth, Figure 5.7 shows the effect of relaxing the harvest constraint at K instead of h_0, thus $0 \leq h \leq K$. This results in more uncertainty being admitted in the damage estimate. In specific, for positive sockeye growth rates, management can have unintended consequences of reducing the benefits of elodea to salmon, thus fisheries will realize forgone benefits. The range of forgone benefits (above the 5th percentile) is equal to the shaded area below the horizontal axis. The median (line) indicates positive net benefits crossing the horizontal zero damage axis in year ten, followed by damages that are similar in magnitude to the no action base-case. Important to note is that NPV is still minimized at t = 1 and does not alter the optimal decision. This result is holds, despite the potential for elodea being a “good” for salmon, the long-term damages outweigh the positive growth effects of elodea on salmon. The probability distribution for θ is responsible for this result (Table 5.4).

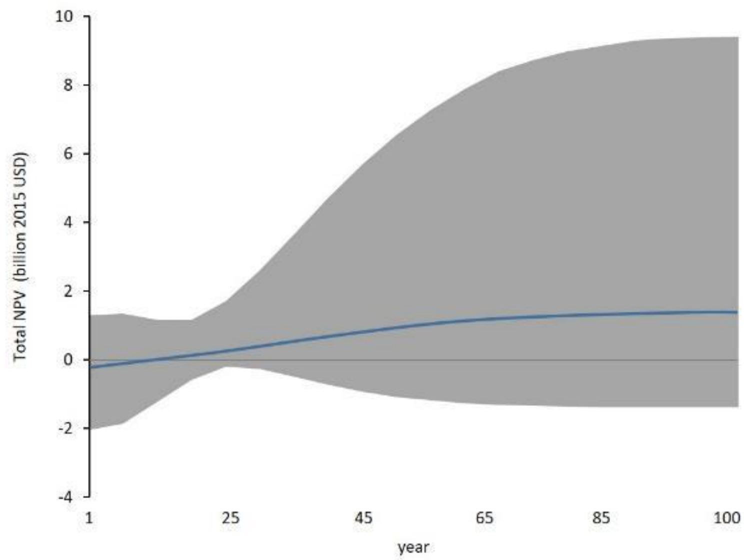

Figure 5.7 NPV of potential damages over time, given annual harvest is constraint by carrying capacity *K* (line is median, shaded is 90% CI).

5.3 Discussion

This study's strategic management recommendations suggest that early and intensive management near the initial invasion site is best rather than later control elsewhere. This result is consistent with other research (Wilen 2007). Especially as agencies face budget limitations, bioeconomic risk analysis can set data-driven priorities that account for social-ecological feedbacks and invasion processes across broad landscapes. The results show that social science can significantly contribute to the parameterization of ecological models, especially if ecological processes are driven by human dispersal, narrowing damage estimates and assisting optimal management. In addition, region-specific estimates of economic values at risk are essential for informing optimal management across large landscapes. The availability of local economic data is often directly linked to whether managers think the model results are reliable and whether local stakeholders trust the estimates. While benefit transfer methods may bridge this gap, they often cannot provide what's needed (Holmes et al. 2010).

In addition, estimating region-specific financial damages to different stakeholders not only illustrates the distribution of damages across ecosystem user groups, but it can create incentives for market-based conservation mechanisms informed by local data (Engel, Pagiola, and Wunder 2008). Private investment in invasive species management in particular can be useful for cases like this one, where resource managers tend to have sole responsibility. This sole responsibility can "crowd out" private investment as evident in Alaska where private funding contributes little to active invasive species management with particular funding gaps for prevention (David Finnoff et al. 2005; Schwörer, Federer, and Ferren 2014).

The sensitivity analysis shows that the damage estimates are robust to parameter assumptions. The logistic growth assumption for the base-case represents a lower bound among the three growth types analysed. The following factors would contribute to higher damages than presented in the base-case. First, the study only includes commercial fisheries impacts to sockeye salmon, excluding all other commercial, subsistence, and sport fisheries for salmon in Alaska. Sockeye salmon catch amounts to 26% of Alaska's commercial salmon catch, that amounts to over half of the value of Alaska's commercial salmon catch (Knapp, Roheim, and Anderson 2007). Since Alaska-specific economic data on the non-market value of subsistence and sport fisheries is rare, the region-specific risk to fisheries is skewed towards regions that have commercial sockeye fisheries. There is evidence that the net economic value of sport and subsistence fisheries can be more than twice as large compared to commercial fisheries (Duffield et al. 2013). The focus on commercial sockeye fisheries particularly affects the Yukon region, where salmon species other than sockeye are a very important resource supporting local subsistence economies (Brown et al. 2015). In addition, under base-case assumptions, the economic risk calculated for the Yukon region is likely underestimated, given unmanaged invasions continue to be elodea sources that eventually may spread into the floatplane vector. Therefore the worst-case which assumes elodea has invaded the regional floatplane hub, may yet provide a better measure of risk and lower bound for this region (Table 5.7).

Second, potential damages to producers are not accounted for, ignoring the income effects to fishermen and the wider economy as well as those related to commercial floatplane operators. Third, the study's approach assumes that the avoided damages through management action are considered a perfect substitute for environmental quality; thus, society can achieve environmental quality through avoidance of the damages. However, this is rarely the case especially when some aspects of avoided cost cannot be offset by management actions (Hanley and Spash 1993). Fourth, damages do not include the effects of elodea on other ecosystem services. For example, there is evidence that elodea affects nutrient cycling (Ozimek, Donk, and Gulati 1993), reduces lakefront property values by up to 16% (Zhang and Boyle 2010), and has severe impacts on biodiversity (Mjelde et al. 2012). This limitation underlines that the true value of ecosystems cannot solely be expressed in monetary units.

Ideally, damage assessments like the one presented would be based on empirical evidence of economic and ecological changes before and after invasions while controlling for different drivers of ecosystem and human system conditions. Such data, would also allow for data-driven validation of the developed model. While the data needs would be enormous, data collection could only occur under experimental settings questioning the validity of the results in practice or after the invasion occurred, making the value of prevention irrelevant. Obtaining SEJ, instead, provides a feasible workaround to data limitation, while being able to explicitly quantify uncertainty in the estimates. Recognizing that SEJ is no panacea for biophysical research that establishes the ecological relationship between the invader and the harvestable resource, SEJ enables researchers to quantify the expected value of information to reduce uncertainty through biophysical experimentation (Peterman and Anderson 1999). Such analysis was outside the scope of this study and serves as a possible extension.

Additional extensions to this research include incorporating a higher resolution approach to spatially explicit risk by refining the underlying species distribution model (SDM). In specific, the floatplane data could be used for developing a probability model associated with the floatplane vector that predicts the probability of colonization in destinations that are currently unknown (Stanaway, Reeves, and Mengersen 2011). The SDM could further include habitat suitability as developed for elodea in Alaska, given absolute probability raster data is available from previous research (Luizza et al. 2016). Integrating a finer-resolution SDM model into the existing analysis could also be used to assess the probability of successfully eradicating elodea from Alaska (Spring and Cacho 2015). While the presented analysis sacrifices some spatial detail it is able to account for important spatial and temporal dynamics affecting several ecosystem services, similar to process-based models that are spatially explicit but the presented approach more easily allows integration of other components such as SDM (Holmes et al. 2010). Despite higher resolution and added capabilities, however, any model will be limited by the number of other vectors it can incorporate. Most importantly, as long as flooding can further distribute elodea across the Yukon region, this region may continue to be a source of elodea for other regions. Since floatplane hubs in the Yukon region are currently elodea free, the presented damages do not fully account for the risk associated with the Yukon region.

6 Conclusion

Upfront management action on aquatic invasive species can have large long-term benefits for the protection of highly productive ecosystems. This study empirically estimates the potential future damages to commercial sockeye salmon fisheries and the potential user loss to recreational floatplane pilots and weighs these damages against the potential cost of management. Results show that if no action is taken to eradicate existing elodea invasions in the state, the median loss of natural capital over a 100-year period amounts to $1,382 million in 2015 USD. The equivalent annual loss in ecosystem services amounts to a median of $50 million (90% CI: $5.5; $263.9). Even though the range of the damage estimate remains large, the median (most probable) estimate suggests that substantial investment is necessary to prevent aquatic invasive species from establishing in Alaska. In this context, establishing funding mechanisms that allow early detection and rapid response is essential.

On a national scale, considering the attention the invasive species threat has received elsewhere but Alaska, this study raises an important point. Bioeconomic research has shown that preventing biological invasions has greater benefits for society compared to managing invasions once they established. This fact raises the question whether past invasive species investments were optimally allocated in ecosystems that will never return to an unimpaired state or whether these investments would be better directed towards preventing damage to some of the most productive ecosystems of national and global significance. In particular, the Bristol Bay region faces the greatest region-specific risk. If no intervention occurs in urban source lakes, there is a greater than one in two chance the Bristol Bay will be invaded now and a one in one chance it will be invaded in the next fifty years. With the invasive species problem in its infancy in Arctic and Subarctic regions, society still has the opportunity to achieve large returns on investment by taking action now, yet, the window of opportunity is quickly closing.

7 References

ACERA. 2010. "Process Manual Elicitation Tool." Melbourne,. http://www.acera.unimelb.edu.au/materials/endorsed/0611-process-manual.pdf.

ADFG. 2016a. "Alaska Commercial Salmon Harvest and Exvessel Value." Anchorage, AK: Alaska Department of Fish and Game. http://www.adfg.alaska.gov/index.cfm%3Fadfg%3DCommercialByFisherySalmon.exvessel query.

———. 2016b. "Commercial Operator's Annual Reports (COAR)." Anchorage, AK: Alaska Department of Fish and Game. http://www.adfg.alaska.gov/index.cfm?adfg=fishlicense.coar.

Adkison, M D, and B P C N - AFRB.10.2.083-094 Finney. 2003. "The Long-Term Outlook for Salmon Returns to Alaska." *Alaska Fishery Research Bulletin* 10 (2): 83–94. http://www.adfg.alaska.gov/FedAidpdfs/AFRB.10.2.083-094.pdf.

Akaike, Hirotugu. 1974. "A New Look at the Statistical Model Identification." *IEEE Transactions on Automatic Control* 19 (6): 716–23. doi:10.1109/TAC.1974.1100705.

Alaska Department of Fish and Game. 2016. "Wildlife Harvest Data." *Harvest Lookup / Data Download*. https://secure.wildlife.alaska.gov/index.cfm?fuseaction=harvest.lookup.

Barko, John W, and William F James. 1998. "Effects of Submerged Aquatic Macrophytes on Nutrient Dynamics, Sedimentation, and Resuspension." In *The Structuring Role of Submerged Macrophytes in Lakes*, edited by Erik Jeppesen, Martin Søndergaard, Morten Søndergaard, and Kirsten Christoffersen, 197–214. Berlin: Springer.

Beattie, Lisa, Christy Everett, Laura Jacobs, Bonnie Million, Cecil Rich, Jeff Rogers, Blaine Spellman, and Tricia L. Wurtz. 2011. "Control Options for Elodea Spp. in the Chena Slough near Fairbanks , Alaska: A Compilation of Potential Treatments." Fairbanks, Alaska. http://aswcd.org/chena_slough_control_options_elodea.pdf.

Ben-Akiva, Moshe, and Steven R. Lerman. 1985. *Discrete Choice Analysis: Theory and Applications to Travel Demand*. Cambridge, MA: MIT Press.

Bowmer, Kathleen H., S.W.L. Jacobs, and G.R. Sainty. 1995. "Identification, Biology and Management of Elodea Canadensis , Hydrocharitaceae." *Journal of Aquatic Plant Management* 33: 13–19.

Brown, Caroline L, Anna Godduhn, Hiroko Ikuta, Brittany Retherford, Alida Trainor, Seth J Wilson, and David Koster. 2015. *Socioeconomic Effects of Declining Salmon Runs on the Yukon River*. Edited by Caroline L. Brown and Anna Godduhn. Fairbanks, AK: Alaska Department of Fish and Game, Division of Subsistence. http://www.adfg.alaska.gov/techpap/TP398.pdf.

Burgman, Mark, James Franklin, Keith R. Hayes, Geoffrey R. Hosack, Gareth W. Peters, and Scott A. Sisson. 2012. “Modeling Extreme Risks in Ecology.” *Risk Analysis* 32 (11): 1956–66. doi:10.1111/j.1539-6924.2012.01871.x.

Burks, Romi L., Erik Jeppesen, and David M. Lodge. 2001. “Littoral Zone Structures as Daphnia Refugia against Fish Predators.” *Limnology and Oceanography* 46 (2): 230–37. doi:10.4319/lo.2001.46.2.0230.

CAFF. 2013. “Arctic Biodiversity Assessment: Status and Trends in Arctic Biodiversity.” Akureyri, IS. http://www.caff.is/.

Carey, Michael, Suresh A. Sethi, Sabrina Larsen, and Cecil Rich. 2016. “A Primer on Potential Impacts, Management Priorities, and Future Directions for Elodea Spp. in High Latitude Systems: Learning from the Alaska Experience.” *Hydrobiologia*. Anchorage, AK: Hy.

Carlson, Matthew L, Irina V. Lapina, and Michael Shephard. 2008. “Invasiveness Ranking System for Non-Native Plants of Alaska.”

Carson, R.T., W.M. Hanemann, and T.C. Wegge. 2009. “A Nested Logit Model of Recreational Fishing Demand in Alaska.” *Marine Resource Economics* 24: 101–30.

Case, Ted J. 2000. “Density Dependent Population Growth.” In *An Illustrated Guide to Theoretical Ecology*, 103–34. Oxford, UK: Oxford University Press.

Casselman, John M, and Cheryl A Lewis. 1996. “Habitat Requirements of Northern Pike (Esox Lucius).” *Canadian Journal of Fisheries and Aquatic Sciences* 53 (1): 161–74.

CH2MHILL. 2005. “Vegetation Management Plan.” Anchorage, AK. http://www.aswcd.org/DOT_AquaticVegetationPlan.pdf.

Champion, Paul D, Mary D De Winton, and John S Clayton. 2014. “A Risk Assessment Based Proactive Management Strategy for Aquatic Weeds in New Zealand.” *Management of Biological Invasions* 5 (3). http://www.reabic.net/journals/mbi/2014/3/MBI_2014_Champion_etal_correctedproof.pdf.

Chen, Heng Z, Frank Lupi, and John P Hoehn. 1997. “An Empirical Assessment of Multinomial Probit and Logit Models for Recreation Demand.” 97–23. Staff Paper. East Lansing, MI. http://ageconsearch.umn.edu/bitstream/201216/2/agecon-msu-97-23.pdf.

Cochran, Roger C. 1994. “Diquat Dibromide Risk Characterization Document.” http://www.cdpr.ca.gov/docs/risk/rcd/diquat.pdf.

Cook, Christopher D.K., and Katharina Urmi-König. 1985. “A Revision of the Genus Elodea (Hydrocharitaceae).” *Aquatic Botany* 21: 111–56.

Cooke, Roger M. 1991. *Experts in Uncertainty: Opinion and Subjective Probability in Science*. New York: Oxford University Press.

Cunningham, Charles E., Ken Deal, and Yvonne Chen. 2010. “Adaptive Choice-Based Conjoint Analysis: A New Patient-Centered Approach to the Assessment of Health Service Preferences.” *Patient* 3 (4): 257–73. doi:10.2165/11537870-000000000-00000.

Davies, P.J., and D.E. Seaman. 1968. “Uptake and Translocation of Diquat in Elodea.” *Weed Science* 16 (3): 293–95.

Deines, Andrew M., Valerie C. Chen, and Wayne G. Landis. 2005. “Modeling the Risks of Nonindigenous Species Introductions Using a Patch-Dynamics Approach Incorporating Contaminant Effects as a Disturbance.” *Risk Analysis* 25 (6): 1637–51. doi:10.1111/j.1539-6924.2005.00680.x.

DeVoretz, D. 1982. “An Econometric Demand Model for Canadian Salmon.” *Canadian Journal of Agricultural Economics* 30 (1): 49–60.

Diehl, S, and R Kornijow. 1998. “Influence of Submerged Macrophytes on Trophic Interactions among Fish and Macroinvertebrates.” *Structuring Role of Subbmerged Macrophytes in Lakes* 131: 24–46.

Diehl, Sebastian, R Kornijow, Ryszard Kornijów, and R Kornijow. 1998. “Influence of Submerged Macrophytes on Trophic Interactions Among Fish and Macroinvertebrates.” In *The Structuring Role of Submerged Macrophytes in Lakes*, edited by Erik Jeppesen, Martin Søndergaard, Morten Søndergaard, and Kirsten Christoffersen, 131:24–46. Berlin: Springer.

Dillman, D.A. 2007. *Mail and Internet Surveys: The Tailored Design Method*. 2nd ed. Hoboken, NJ: Joihn Wiley & Sons, Inc.

Ding, Min, and Joel Huber. 2009. “When Is Hypothetical Bias a Problem in Choice Tasks, and What Can We Do about It?” In *Proceedings of the Sawtooth Software Conference*, 263–72. Sawtooth Software, Inc. http://www.sawtoothsoftware.com/download/techpap/2009Proceedings.pdf.

DNR. 2015. “Anchorage Lakes Elodea Eradication Project: Environmental Assessment.” Anchorage, AK. http://plants.alaska.gov/invasives/pdf/FINAL_EA_Anchorage2015.pdf.

Doelle, Meinhard. 2003. “The Quiet Invasion : Legal and Policy Responses to Aquatic Invasive Species in North America.” *International Journal of Marine and Coastal Law* 18 (2): 261–95.

Drescher, M, A H Perera, C J Johnson, L J Buse, and C A Drew. 2013. “Toward Rigorous Use of Expert Knowledge in Ecological Research.” *Ecosphere* 4 (7): Article 83.

Drew, Ashton, and Ajith H. Perera. 2011. “Predictive Species and Habitat Modeling in Landscape Ecology: Concepts and Applications.” In *Predictive Species and Habitat Modeling in Landscape Ecology: Concepts and Applications*, edited by C. Ashton Drew, Yolanda F. Wiersma, and Falk Huettmann, 1–313. Springer. doi:10.1007/978-1-4419-7390-0.

Duffield, John, Chris Neher, David Patterson, Gunnar Knapp, Tobias Schwörer, Ginny Fay, and

Oliver Scott Goldsmith. 2013. “Bristol Bay Wild Salmon Ecosystem: Baseline Levels of Economic Activity and Values.” *An Assessment of Potential Mining Impacts on Salmon Ecosystems of Bristol Bay, Alaska*. U.S. Environmental Protection Agency. http://cfpub.epa.gov/ncea/bristolbay/recordisplay.cfm?deid=253500#Download.

Edmundson, J A, and A Mazumder. 2001. “Linking Growth of Juvenile Sockeye Salmon to Habitat Temperature in Alaskan Lakes.” *Transactions of the American Fisheries Society* 130 (4): 644–62. doi:10.1577/1548-8659(2001)130.

Engel, Stefanie, Stefano Pagiola, and Sven Wunder. 2008. “Designing Payments for Environmental Services in Theory and Practice: An Overview of the Issues.” *Ecological Economics* 65 (4): 663–74. doi:10.1016/j.ecolecon.2008.03.011.

Epanchin-Niell, Rebecca S., and Alan Hastings. 2010. “Controlling Established Invaders: Integrating Economics and Spread Dynamics to Determine Optimal Management.” *Ecology Letters* 13: 528–41. doi:10.1111/j.1461-0248.2010.01440.x.

Erhard, Daniela, Georg Pohnert, and Elisabeth M. Gross. 2007. “Chemical Defense in Elodea Nuttallii Reduces Feeding and Growth of Aquatic Herbivorous Lepidoptera.” *Journal of Chemical Ecology* 33 (8): 1646–61. doi:10.1007/s10886-007-9307-0.

FAA. 2015. “Airmen Certification Database.” Federal Aviation Administration. http://www.faa.gov/licenses_certificates/airmen_certification/releasable_airmen_download/.

Facon, Benoît, and Patrice David. 2006. “Metapopulation Dynamics and Biological Invasions: A Spatially Explicit Model Applied to a Freshwater Snail.” *The American Naturalist* 168 (6): 769–83. doi:10.1086/508669.

Finnoff, D., C. McIntosh, Jason F. Shogren, Charles Sims, and Travis W. Warziniack. 2010. “Invasive Species and Endogenous Risk.” *Annual Review of Resource Economics* 2: 77–100.

Finnoff, David, Jason F. Shogren, Brian Leung, and David Lodge. 2005. “The Importance of Bioeconomic Feedback in Invasive Species Management.” *Ecological Economics* 52 (3): 367–81. doi:10.1016/j.ecolecon.2004.06.020.

———. 2007. “Take a Risk: Preferring Prevention over Control of Biological Invaders.” *Ecological Economics* 62 (2): 216–22. doi:10.1016/j.ecolecon.2006.03.025.

Franklin, James, Scott a. Sisson, Mark a. Burgman, and Jennifer K. Martin. 2008. “Evaluating Extreme Risks in Invasion Ecology: Learning from Banking Compliance.” *Diversity and Distributions* 14 (4): 581–91. doi:10.1111/j.1472-4642.2008.00478.x.

Freeman, A Myrick. 2003. *The Measurement of Environmental and Resource Values*. 2nd ed. Washington, D.C.: Resources for the Future.

Friedman, Sam. 2015. “Invasive Elodea Found North of Nenana; First Interior Sighting beyond

Fairbanks Area.” *Fairbanks Daily News Miner*, September 1. http://www.newsminer.com/news/local_news/invasive-elodea-found-north-of-nenana-first-interior-sighting-beyond/article_cd427cf0-511d-11e5-8c60-c733aa6798ec.html.

Gelman, Andrew, John B. Carlin, Hal S. Stern, David B. Dunson, Aki Vehtari, and Donald B. Rubin. 2013. *Bayesian Data Analysis*. New York: Chapman & Hall/CRC.

Glover, F., J.P. Kelly, and M. Laguna. 1996. “New Advances and Applications of Combining Simulation and Optimization.” In *Proceedings of the 1996 Winter Simulation Conference*, edited by J.M. Charnes, D.J. Morrice, D.T. Brunner, and J.J. Swain, 144–52.

Halstead, John M. J.M., Jodi Michaud, Shanna Hallas-Burt, and Julie P. Gibbs. 2003. “Hedonic Analysis of Effects of a Nonnative Invader (Myriophyllum Heterophyllum) on New Hampshire (USA) Lakefront Properties.” *Environmental Management* 32 (3): 391–98. doi:10.1007/s00267-003-3023-5.

Hamelink, Jerry L., Denny R. Buckler, Foster L. Mayer, Donald U. Palawski, and Herman O. Sanders. 1986. “Toxicity of Fluridone to Aquatic Invertebrates and Fish.” *Environmental Toxicology and Chemistry* 5: 87–94.

Hanley, Nick, and Clive L. Spash. 1993. *Cost-Benefit Analysis and the Environment*. Northampton, MA: Edward Elgar Publishing Limited.

Hanley, Nick, Robert E Wright, and Gary Koop. 2002. “Modelling Recreation Demand Using Choice Experiments : Climbing in Scotland.” *Environmental and Resource Economics* 22: 449–66.

Hare, S.R., N.J. Mantua, and R.C. Francis. 1999. “Inverse Production Regimes: Alaska and West Coast Pacific Salmon.” *Fisheries* 24: 6–14.

Harper, Peter, Bill Chisholm, and Nimal Chandrasena. 2007. “Herbicide Trials for the Control of Submerged Aquatic Weeds.” North Otago, New Zealand. http://www.chisholm.co.nz/Documents/WoolongongConf_Hydrogel_Aquatic_Weeds_Final_June2007.pdf.

Hausman, Jerry A, Gregory K Leonard, and Daniel McFadden. 1995. “A Utility-Consistent, Combined Discrete Choice and Count Data Model: Assessing Recreational Use Losses due to Natural Resource Damage.” *Journal of Public Economics* 56: 1–30.

Hausman, Jerry A, and David A Wise. 1978. “A Conditional Probit Model for Qualitative Choice: Discrete Decisions Recognizing Interdependence and Heterogeneous Preferences.” *Econometrica* 46 (2): 403–26.

Heikkinen, Risto, Niko Leikola, Stefan Fronzek, Raino Lampinen, and Heikki Toivonen. 2009. “Predicting Distribution Patterns and Recent Northward Range Shift of an Invasive Aquatic Plant: Elodea Canadensis in Europe.” *BioRisk* 2 (November): 1–32. doi:10.3897/biorisk.2.4.

Hensher, David A., John M. Rose, and William H. Greene. 2005. *Applied Choice Analysis: A Primer*. Cambridge, UK: Cambridge University Press.

Hiebert, Ronald D., and James Stubbendieck. 1993. “Handbook for Ranking Exotic Plants for Management and Control.” Lincoln, NE. http://especes-envahissantes-outremer.fr/pdf/methode_hierarchisation_hiebert.pdf.

Hoehn, John P., Frank Lupi, and Michael D. Kaplowitz. 2010. “Stated Choice Experiments with Complex Ecosystem Changes: The Effect of Information Formats on Estimated Variances and Choice Parameters.” *Journal of Agricultural and Resource Economics* 35 (3): 568–90.

Hole, Arne Risa. 2007. “A COMPARISON OF APPROACHES TO ESTIMATING CONFIDENCE INTERVALS FOR WILLINGNESS TO PAY MEASURES.” *Health Economics* 840 (January): 827–40. doi:10.1002/hec.

Hollander, Zaz. 2014. “Lake - Choking Invasive Weed Makes It to Mat -Su.” *Alaska Dispatch News*, October 16. http://www.adn.com/article/20141016/lake-choking-invasive-weed-makes-it-mat-su.

———. 2015a. “Weed Killer Proposed for Invasive Plant Choking West Anchorage Lakes.” *Alaska Dispatch News*, June 6. http://www.adn.com/article/20150606/weed-killer-proposed-invasive-plant-choking-west-anchorage-lakes.

———. 2015b. “Pilots Push for Weed Killer to Combat Lake Hood’s Plane-Fouling Plants.” *Alaska Dispatch News*, July 10. http://www.adn.com/article/20150710/pilots-push-weed-killer-combat-lake-hood-s-plane-fouling-plants.

Hollister, Jeffrey W. 2016. “Lakemorpho: Lake Morphometry Metrics in R.” R-Project. https://cran.r-project.org/package=lakemorpho.

Holmes, Thomas P, Andrew M Liebhold, Kent F Kovacs, and Betsy Von Holle. 2010. “A Spatial-Dynamic Value Transfer Model of Economic Losses from a Biological Invasion.” *Ecological Economics* 70 (1). Elsevier B.V.: 86–95. doi:10.1016/j.ecolecon.2010.07.035.

Horsch, Eric J, and David J Lewis. 2008. “The Effects of Aquatic Invasive Species on Property Values: Evidence from a Quasi-Random Experiment.” 530. *Agricultural & Applied Economics Staff Paper Series*. Madison, WI.

Howell, John. 2007. “Respondent Weighting in HB.” In *Proceedings of the Sawtooth Software Conference*, 365–77. Santa Rosa, CA: Sawtooth Software. http://www.sawtoothsoftware.com/download/techpap/2007Proceedings.pdf.

Hoyos, David. 2010. “The State of the Art of Environmental Valuation with Discrete Choice Experiments.” *Ecological Economics* 69 (8). Elsevier B.V.: 1595–1603. doi:10.1016/j.ecolecon.2010.04.011.

Hussner, A, I Stiers, M J J M Verhofstad, E S Bakker, B M C Grutters, J Haury, G van Valkenburg, J L CH Brundu, et al. 2017. “Management and Control Methods of Invasive Alien Freshwater Aquatic Plants: A Review.” *Aquatic Botany* 136. Elsevier B.V.: 112–37. doi:10.1016/j.aquabot.2016.08.002.

Ishizaka, Alessio, and Ashraf Labib. 2009. “Analytic Hierarchy Process and Expert Choice: Benefits and Limitations.” *OR Insight* 22 (4). Palgrave Macmillan UK: 201–20. doi:10.1057/ori.2009.10.

Jeppesen, Erik, Torben L. Lauridsen, Timo Kairesalo, and Martin R. Perrow. 1998. “Impact of Submerged Macrophytes on Fish-Zooplankton Interactions in Lakes.” In *The Structuring Role of Submerged Macrophytes in Lakes*, edited by Erik Jeppesen, Martin Søndergaard, Morten Søndergaard, and Kirsten Christoffersen, 91–114. Berlin. doi:10.1007/978-1-4612-0695-8_5.

Jeschke, Jonathan M, Sven Bacher, T I M M Blackburn, Jaimie T A Dick, Franz Essl, Thomas Evans, Mirijam Gaertner, et al. 2014. “Defining the Impact of Non-Native Species.” *Conservation Biology* 28 (5): 1188–94. doi:10.1111/cobi.12299.

Johnson, Rich, Joel Huber, and Lynd Bacon. 2003. “Adaptive Choice-Based Conjoint.” *Sawtooth Software Research Paper Series*. Vol. 98382. http://www.sawtoothsoftware.com/download/techpap/acbc.pdf.

Johnson, Rick. 2013. “AQUATIC WEEDS MANAGEMENT FUND FINAL REPORT Chehalis River.” http://www.co.thurston.wa.us/tcweeds/docs/final report elodeaecy13.pdf.

Johnston, Richard S., and W. Robert Wood. 1974. “Demand Analysis for Canned Red (Sockeye) Salmon at Wholesale.” Corvallis, OR. http://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/4522/?sequence=1.

Johnstone, I.M. M, B.T. T Coffey, and C. Howard-Williams. 1985. “The Role of Recreational Boat Traffic in Interlake Dispersal of Macrophytes: A New Zealand Case Study.” *Journal of Environmental Management* 20 (3): 263–79. http://www.scopus.com/inward/record.url?eid=2-s2.0-0022236188&partnerID=40&md5=b4f074cb43a04d15f8d84a247b8006a1.

Josefsson, Melanie. 2011. “Invasive Alien Species Fact Sheet Elodea Canadensis , Elodea Nuttallii and Elodea Callitrichoides.” Stockholm, Sweden: Department of Natural Resources, Swedish Environmental Protection Agency. http://www.nobanis.org/files/factsheets/Elodea.pdf.

Kamarianos, A., J. Altiparmakis, X. Karamanlis, D. Kufidis, T. Kousouris, G. Fotis, and S. Kilikidis. 1989. “Experimental Evaluation of Fluridone Effectiveness on Fish Productive Aquatic Ecosystems.” *Journal of Aquatic Plant Management* 27: 24–26.

Knapp, Gunnar, Mouhcine Guettabi, and Oliver Scott Goldsmith. 2013. “The Economic Importance of the Bristol Bay Salmon Industry.” Anchorage, AK.

Knapp, Gunnar, Cathy A. Roheim, and James L. Anderson. 2007. *The Great Salmon Run: Competition between Wild and Farmed Salmon*. Washington, D.C.: TRAFFIC North America World Wildlife Fund.

Knowler, D., N. Philcox, S. Nathan, W. Delamare, W. Haider, and K. Gupta. 2009. "Assessing Prospects for Shrimp Culture in the Indian Sundarbans: A Combined Simulation Modelling and Choice Experiment Approach." *Marine Policy* 33 (4): 613–23. doi:10.1016/j.marpol.2008.12.009.

Krinsky, Itzhak, and A Leslie Robb. 1986. "On Approximating the Statistical Properties of Elasticities." *The Review of Economics and Statistics* 68 (4): 715–19.

Lane, Ryan A. 2014. "Chena Slough Elodea Control Trial Project: 2013 Overview." Fairbanks, AK.

Levins, Richard. 1969. "Some Demographic and Genetic Consequences of Environmental Heterogeneity for Biological Control." *Bulletin of the Entomological Society of America* 15 (3): 237–240. doi:10.1093/besa/15.3.237.

Lodge, David M, Paul W Simonin, Stanley W Burgiel, Reuben P Keller, Jonathan M Bossenbroek, Christopher L Jerde, Andrew M Kramer, et al. 2016. "Risk Analysis and Bioeconomics of Invasive Species to Inform Policy and Management." *Annual Review of Environment and Resources* 41: 453–88. doi:10.1146/annurev-environ-110615-085532.

Louviere, Jordan J., David A. Hensher, and Joffre Swait. 2000. *State Choice Models: Analysis and Applications in Marketing, Transportation and Environmental Valuation*. Cambridge, MA: Cambridge University Press.

Louviere, Jordan J, Nada Wasi, Deborah Street, Towhidul Islam, and Anthony A J Marley. 2007. "Modeling the Choices of Individual Decision-Makers by Combining Efficient Choice Experiment Designs with Extra Preference Information." *Journal of Choice Modelling* 1 (1): 128–63.

Low-Choy, Samantha, Rebecca O'Leary, and Kerrie Mengersen. 2009. "Elicitation by Design in Ecology: Using Expert Opinion to Inform Priors for Bayesian Statistical Models." *Ecology* 90 (1): 265–77. doi:10.1890/07-1886.1.

Luizza, Matthew W, Paul H Evangelista, Catherine S Jarnevich, Amanda West, and Heather Stewart. 2016. "Integrating Subsistence Practice and Species Distribution Modeling: Assessing Invasive Elodea's Potential Impact on Native Alaskan Subsistence of Chinook Salmon and Whitefish." *Environmental Management*. Springer US. doi:10.1007/s00267-016-0692-4.

Maguire, Lynn A. 2004. "What Can Decision Analysis Do for Invasive Species Management?" *Risk Analysis* 24 (4): 859–68. doi:10.1111/j.0272-4332.2004.00484.x.

Manski, Charles F, and Daniel McFadden. 1981. *Structural Analysis of Discrete Data with*

Econometric Applications. Cambridge, MA: MIT Press.

Martin, Tara G, Mark a Burgman, Fiona Fidler, Petra M Kuhnert, Samantha Low-Choy, Marissa McBride, and Kerrie Mengersen. 2012. "Eliciting Expert Knowledge in Conservation Science." *Conservation Biology* 26 (1): 29–38. doi:10.1111/j.1523-1739.2011.01806.x.

McDowell Group. 2015. "Sockeye Market Analysis." Juneau, AK. http://www.mcdowellgroup.net/wp-content/uploads/2015/12/Fall-2015-Sockeye-Market-Report-Final.pdf.

McElhany, P., Mary H. Rucklelshaus, Michael J. Ford, Thomas C. Wainwright, and Eric Bjorkstedt. 2000. "Viable Salmonid Populations and the Recovery of Evolutionarily Significant Units." *NOAA Technical Memorandum NMFS-NWFSC ; 42.*, no. June: 156. http://www.nwfsc.noaa.gov/assets/25/6190_06162004_143739_tm42.pdf.

McFadden, Daniel. 1973. "Conditional Logit Analysis of Qualitative Choice Behavior." In *Frontiers in Econometrics*, edited by P. Zarembka, 105–42. New York: Academic Press. doi:10.1108/eb028592.

Merz, Joseph E., James R. Smith, Michelle L. Workman, Jose D. Setka, and Bert Mulchaey. 2008. "Aquatic Macrophyte Encroachment in Chinook Salmon Spawning Beds: Lessons Learned from Gravel Enhancement Monitoring in the Lower Mokelumne River, California." *North American Journal of Fisheries Management* 28 (5): 1568–77. doi:10.1577/M07-043.1.

Mjelde, Marit, Paola Lombardo, Dag Berge, and Stein W. Johansen. 2012. "Mass Invasion of Non-Native Elodea Canadensis Michx. in a Large, Clear-Water, Species-Rich Norwegian Lake – Impact on Macrophyte Biodiversity." *Annales de Limnologie - International Journal of Limnology* 48 (2): 225–40. doi:10.1051/limn/2012016.

Morgan, M. Granger, and Max Henrion. 1990. *Uncertainty: A Guide to Dealing with Uncertainty*. Cambridge, UK: Cambridge University Press.

Morgan, M Granger. 2014. "Use (and Abuse) of Expert Elicitation in Support of Decision Making for Public Policy." *Proceedings of the National Academy of Sciences of the United States of America* 111 (20): 7176–84. doi:10.1073/pnas.1319946111.

Morton, John. 2016. "Refuge Notebook : Elodea Gone from the Kenai Peninsula ?" *Peninsula Clarion*, June 2.

Morton, John M., Brianne N. Blackburn, Elizabeth Bella, Matt Steffy, Cheryl Anderson, Rob Massengil, Jack Blackwell, et al. 2014. "Integrated Pest Management Plan for Eradicating Elodea From the Kenai Peninsula." Kenai, Alaska. http://www.fws.gov/uploadedFiles/Integrated_Pest_Management_Plan_for_Eradicating_Elodea_ver3_1.pdf.

Nawrocki, Timm, Helen I Klein, Matthew L Carlson, Jeff Conn, Whitney Rapp, Lindsey A Flagstad, Gino Graziano, and Robert L Develice. 2011. "Alaska Non-Native Plant Invasiveness

Ranking Form: Elodea Canadensis Michx." Anchorage, AK: Alaska Natural Heritage Program. http://aknhp.uaa.alaska.edu/services/aknhp/aknhp.cfc?method=downloadDocumentByUsdaCode&documentType=risk_assessment&usdaCode=ELODE.

Nunes, Paulo a.L.D., and Jeroen C.J.M. van den Bergh. 2004. "Can People Value Protection against Invasive Marine Species? Evidence from a Joint TC–CV Survey in the Netherlands." *Environmental & Resource Economics* 28 (4): 517–32. doi:10.1023/B:EARE.0000036777.83060.b6.

Olden, Julian D, and Mariana Tamayo. 2014. "Incentivizing the Public to Support Invasive Species Management: Eurasian Milfoil Reduces Lakefront Property Values." *PloS One* 9 (10): e110458. doi:10.1371/journal.pone.0110458.

OMB. 2016. "2017 Discount Rates for OMB Circular No. A-94." Washington, DC: U.S. Executive Office of the President Office of Management and Budget. https://www.whitehouse.gov/sites/default/files/omb/memoranda/2017/m-17-10.pdf.

Orme, Bryan. 2009a. "Fine-Tuning CBC and Adaptive CBC Questionnaires." *Technical Paper Available at Www. Sawtoothsoftware. …* 98382 (360). http://www.skimanalytical.nl/images/stories/technicalpapers/CBC-related papers/finetune.pdf.

———. 2009b. "The CBC/HB System for Hierarchical Bayes Estimation." *Sawtooth Software Technical Paper Series*, 1–27. http://www.sawtoothsoftware.com/support/technical-papers/hierarchical-bayes-estimation/cbc-hb-technical-paper-2009.

———. 2010. "Interpreting the Results of Conjoint Analysis." In *Getting Started with Conjoint Analysis: Strategies for Product Design and Pricing Research*, 77–88. Madison, WI: Research Publishers LLC. http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Interpreting+the+Results+of+Conjoint+Analysis#0.

Ozimek, Teresa, Ellen Van Donk, and Ramesh D Gulati. 1993. "Growth and Nutrient Uptake by Two Species of Elodea in Experimental Conditions and Their Role in Nutrient Accumulation in a Macrophyte- Dominated Lake." *Hydrobiologia* 251 (1–3): 13–18.

Palisade. 2016. "@Risk Risk Analysis Using Monte Carlo Simulation." Ithaca, NY: Palisade Corporation.

Palisade Corporation. 2016. "@RISK: Risk Analysis and Simulation Add-in for Microsoft® Excel." Ithaca, NY: Palisade Corporation.

Peterman, Randall M., and Judith L. Anderson. 1999. "Decision Analysis: A Method for Taking Uncertainties into Account in Risk-Based Decision Making." *Human and Ecological Risk Assessment: An International Journal* 5 (2): 231–44. doi:10.1080/10807039991289383.

Peterson, Douglas P, Bruce E Rieman, Jason B Dunham, Kurt D Fausch, and Michael K Young. 2008. “Analysis of Trade-Offs between Threats of Invasion by Nonnative Brook Trout (Salvelinus Fontinalis) and Intentional Isolation for Native Westslope Cutthroat Trout (Oncorhynchus Clarkii Lewisi).” *Canadian Journal of Fisheries and Aquatic Sciences* 65 (4): 557–73. doi:10.1139/f07-184.

Pheloung, P C, P A Williams, and S R Halloy. 1999. “A Weed Risk Assessment Model for Use as a Biosecurity Tool Evaluating Plant Introductions.” *Journal of Environmental Management* 57 (April): 239–51.

Pimentel, David. 2005. “Environmental and Economic Costs of the Application of Pesticides Primarily in the United States In Integrated Pest Management: Innovation-Development Process.” *Environment, Development and Sustainability* 7: 229–52. doi:10.1007/978-1-4020-8992-3_4.

Redekop, Petra, Deborah Hofstra, and Andreas Hussner. 2016. “Elodea Canadensis Shows a Higher Dispersal Capacity via Fragmentation than Egeria Densa and Lagarosiphon Major.” *Aquatic Botany* 130. Elsevier B.V.: 45–49. doi:10.1016/j.aquabot.2016.01.004.

Regan, Helen M, Yakov Ben-Haim, Bill Langford, William G Wilson, Per Lundberg, Sandy J Andelman, and Mark A Burgman. 2005. “Robust Decision-Making under Severe Uncertainty for Conservation Management.” *Ecological Applications* 15 (4): 1471–77.

Riis, Tenna, and Barry J. F. Biggs. 2003. “Hydrologic and Hydraulic Control of Macrophyte Establishment and Performance in Streams.” *Limnology and Oceanography* 48 (4): 1488–97. doi:10.4319/lo.2003.48.4.1488.

Rinella, Daniel J., Daniel L. Bogan, Samuel Call, and James Willacker. 2008. “2008 Cook Inlet Basin Lakes Survey Summary Report.” Anchorage, AK. http://dec.alaska.gov/water/wqsar/monitoring/documents/Final-2008CookInletLakes.pdf.

Rørslett, Bjørn, D A G Berge, and Steine W. Johansen. 1986. “Lake Enrichment by Submersed Macrophytes: A Norwegian Whole-Lake Experience with Elodea Canadensis.” *Aquatic Botany* 26: 325–40.

Rothlisberger, John D, David C Finnoff, Roger M Cooke, and David M Lodge. 2012. “Ship-Borne Nonindigenous Species Diminish Great Lakes Ecosystem Services.” *Ecosystems* 15: 462–76. doi:10.1007/s10021-012-9522-6.

Saaty, Thomas L. 1990. “How to Make a Decision: The Analytic Hierarchy Process.” *European Journal of Operations Research* 48: 9–26.

Sawtooth. 2016a. “Choice Simulator.” Sawtooth Software, Inc.

———. 2016b. “Lighthouse Studio.” Sawtooth Software, Inc.

Schardt, Jeffrey D. 2014. “Annual Report of Activities Conducted under the Cooperative Aquatic

Plant Control Program in Florida Public Waters for Fiscal Year 2008-2009.” Vol. 22. Tallahassee, FL. http://plants.ifas.ufl.edu/manage/sites/default/files/fwcaquaticplantcontrolprogramannualreportfy08-09.pdf.

Schindler, Daniel E, Ray Hilborn, Brandon Chasco, Christopher P Boatright, Thomas P Quinn, Lauren a Rogers, and Michael S Webster. 2010. “Population Diversity and the Portfolio Effect in an Exploited Species.” *Nature* 465 (June). Nature Publishing Group: 609–12. doi:10.1038/nature09060.

Schneider, James C. 2000. “Evaluation of the Effects of the Herbicide Sonar on Sport Fish Populations in Michigan Lakes.” 2000–2. Michigan Department of Natural Resources Fisheries Technical Report. Ann Arbor, MI. http://www.aswcd.org/MDEQstudy-Sonar-on-fish-population.pdf.

Schreiber, Sebastian J, and James O Lloyd-Smith. 2009. “Invasion Dynamics in Spatially Heterogeneous Environments.” *The American Naturalist* 174 (4): 490–505. doi:10.1086/605405.

Schultz, Rachel, and Eric Dibble. 2012. “Effects of Invasive Macrophytes on Freshwater Fish and Macroinvertebrate Communities: The Role of Invasive Plant Traits.” *Hydrobiologia* 684 (1): 1–14. doi:10.1007/s10750-011-0978-8.

Schwörer, Tobias, Rebekka N Federer, and Howard J Ferren. 2014. “Invasive Species Management Programs in Alaska : A Survey of Statewide Expenditures , 2007 – 11.” *Arctic* 67 (1): 20–27. doi:http://dx.doi.org/10.14430/arctic4359.

Sepulveda, Adam J., David S. Rutz, Aaron W. Dupuis, Patrick a. Shields, and Kristine J. Dunker. 2014. “Introduced Northern Pike Consumption of Salmonids in Southcentral Alaska.” *Ecology of Freshwater Fish*, July, n/a-n/a. doi:10.1111/eff.12164.

Sepulveda, Adam J., David S. Rutz, Sam S. Ivey, Kristine J. Dunker, and Jackson a. Gross. 2013. “Introduced Northern Pike Predation on Salmonids in Southcentral Alaska.” *Ecology of Freshwater Fish* 22 (2): 268–79. doi:10.1111/eff.12024.

Shaw, W Douglass, and Richard T Woodward. 2008. “Why Environmental and Resource Economists Should Care about Non-Expected Utility Models.” *Resource and Energy Economics* 30 (1): 66–89. doi:10.1016/j.reseneeco.2007.05.001.

Shogren, J.F., David Finnoff, Chris McIntosh, and Chad Settle. 2006. “Integration-Valuation Nexus in Invasive Species Policy.” *Agricultural and Resource Economics Review* 35 (1): 11–20.

Shogren, Jason F. 2000. “Risk Reduction Strategies against the ‘Explosive Invader.’” In *The Economics of Biological Invasions*, edited by Charles Perrings, Mark Williamson, and Silvana Dalmazzone, 56–69. Northampton, MA: Edward Elgar.

Sikder, Iftikhar U., Sanchita Mal-Sarkar, and Tarun K. Mal. 2006. “Knowledge-Based Risk Assessment under Uncertainty for Species Invasion.” *Risk Analysis* 26 (1): 239–52. doi:10.1111/j.1539-6924.2006.00714.x.

Simberloff, Daniel, and Betsy Von Holle. 1999. “Positive Interactions of Nonindigenous Species: Invasional Meltdown?” *Biological Invasions*, 21–32. doi:10.1023/a:1010086329619.

Simpson, D. A. 1984. “A Short History of the Introduction and Spread of Elodea Michx in the British Isles.” *Watsonia* 15: 1–9.

Sinnott, Rick. 2013. “With a Noxious Aquatic Weed Threatening Alaska’s Prime Fishing Waters, Science and Local Politics Are at Odds.” *Alaska Dispatch*, November 28. http://www.adn.com/article/20131128/noxious-aquatic-weed-threatening-alaskas-prime-fishing-waters-science-and-local.

———. 2014. “Alternate Plans to Rid Sand Lake of Noxious Weed Are Nonsense.” *Alaska Dispatch News*, June 22. http://www.adn.com/article/20140622/alternate-plans-rid-sand-lake-noxious-weed-are-nonsense.

Speirs-Bridge, Andrew, Fiona Fidler, Marissa McBride, Louisa Flander, Geoff Cumming, and Mark Burgman. 2010. “Reducing Overconfidence in the Interval Judgments of Experts.” *Risk Analysis* 30 (3): 512–23. doi:10.1111/j.1539-6924.2009.01337.x.

Spicer, K.W., and Paul M. Catling. 1988. “The Biology of Canadian Weeds. 88.” *Canadian Journal of Plant Science* 68 (4): 1035–51.

Spring, Daniel, and Oscar J Cacho. 2015. “Estimating Eradication Probabilities and Trade-Offs for Decision Analysis in Invasive Species Eradication Programs.” *Biological Invasions* 17: 191–204. doi:10.1007/s10530-014-0719-9.

Stanaway, M.A., R. Reeves, and K.L. Mengersen. 2011. “Hierarchical Bayesian Modelling of Plant Pest Invasions with Human-Mediated Dispersal.” *Ecological Modelling* 222 (19): 3531–40. doi:10.1016/j.ecolmodel.2011.08.013.

State of Alaska. 2016. *Plant Quarantine Summary*. http://hdl.handle.net/10900/49803.

Steiner, Erin M., Keith R. Criddle, and Milo D. Adkison. 2011. “Balancing Biological Sustainability with the Economic Needs of Alaska’s Sockeye Salmon Fisheries.” *North American Journal of Fisheries Management* 31 (3): 431–44. doi:10.1080/02755947.2011.588917.

Stewart, Heather. 2016. “Personal Communication.” Palmer, AK: Alaska Department of Natural Resources.

Strecker, Angela L., Philip M. Campbell, and Julian D. Olden. 2011. “The Aquarium Trade as an Invasion Pathway in the Pacific Northwest.” *Fisheries* 36 (2): 74–85. doi:10.1577/03632415.2011.10389070.

Swartz, A. Nelson. 1978. "The Pacific Coho Salmon Fishery: An Intraseasonal and Interregional Economic Analysis of the Ex-Vessel Market." Oregon State University. https://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/25951/SWARTZANELSON1979.pdf?sequence=1.

Sytsma, Mark D, and Toni Pennington. 2015. "Vectors for Spread of Invasive Freshwater Vascular Plants with a North American Analysis." In *Biological Invasions in Changing Ecosystems Vectors, Ecological Impacts, Management and Predictions*, edited by João Canning-Clode, 55–74. Berlin: Walter de Gruyter GmbH & Co KG.

Train, Kenneth E. 2003. *Discrete Choice Methods with Simulation*. *Cambridge University Press*. Cambridge, UK: Cambridge University Press. doi:10.1017/CBO9780511753930.

Turner, R. K., and G. C. Daily. 2007. "The Ecosystem Services Framework and Natural Capital Conservation." *Environmental and Resource Economics* 39 (1): 25–35. doi:10.1007/s10640-007-9176-6.

Tversky, A, and D Kahneman. 1974. "Judgment under Uncertainty: Heuristics and Biases." *Science (New York, N.Y.)* 185 (4157): 1124–31. doi:10.1126/science.185.4157.1124.

U.S. Census Bureau. 2014. "American Community Survey."

U.S. Fish and Wildlife Service, and U.S. Census Bureau. 2013. "2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation." http://www.census.gov/prod/2012pubs/fhw11-nat.pdf.

USGS. 2016. "National Hydrography Dataset." http://nhd.usgs.gov/data.html.

von Gaudecker, Hans Martin, Arthur van Soest, and Erik Wengström. 2012. "Experts in Experiments: How Selection Matters for Estimated Distributions of Risk Preferences." *Journal of Risk and Uncertainty* 45 (2): 159–90. doi:10.1007/s11166-012-9151-7.

Wang, Hsiung Der. 1976. "An Econometric Study of the Canadian Sockeye Salmon Market." Oregon State University.

Warner, P.J., C.C. Bossard, M.L. Brooks, J.M. DiTomaso, J.A. Hall, A.M. Hawald, D.W. Johnson, et al. 2003. "Criteria for Categorizing Invasive Non-Native Plants That Threaten Wildlands." California Exitic Pest Plant Council. http://www.cal-ipc.org/ip/inventory/pdf/Criteria.pdf.

White, Halbert. 1980. "A Heteroskedasticity-Consistent Covariance Matrix Estimator and a Direct Test for Heteroskedasticity." *Econometrica* 48 (4): 817–38.

Wibbenmeyer, Matthew J, Michael S Hand, David E Calkin, Tyron J Venn, and Matthew P Thompson. 2013. "Risk Preferences in Strategic Wildfire Decision Making: A Choice Experiment with U.S. Wildfire Managers." *Risk Analysis* 33 (6): 1021–37. doi:10.1111/j.1539-6924.2012.01894.x.

Willis, Henry H., Michael L. DeKay, M. Granger Morgan, H. Keith Florig, and Paul S. Fischbeck. 2004. "Ecological Risk Ranking: Development and Evaluation of a Method for Improving Public Participation in Environmental Decision Making." *Risk Analysis* 24 (2): 363–78. doi:10.1111/j.0272-4332.2004.00438.x.

Yamada, Kuniko, Jane Elith, Michael McCarthy, and Andre Zerger. 2003. "Eliciting and Integrating Expert Knowledge for Wildlife Habitat Modelling." *Ecological Modelling* 165 (2–3): 251–64. doi:10.1016/S0304-3800(03)00077-2.

Zhang, Congwen, and Kevin J. Boyle. 2010. "The Effect of an Aquatic Invasive Species (Eurasian Watermilfoil) on Lakefront Property Values." *Ecological Economics* 70 (December). Elsevier B.V.: 394–404. doi:10.1016/j.ecolecon.2010.09.011.