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Natural resource system size can be used for managing recreational use

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A B S T R A C T

Outdoor recreation provides societal benefits that are often measured by the amount of use natural resource systems receive. Still, the amount of resource use natural resource systems receive is often unknown or unstudied. Monitoring and quantifying resource use is often logistically difficult and costly but is paramount to optimize societal benefits. Identifying a simple and readily available metric that can indicate the quantity of recreational use of natural resource systems would benefit natural resource management. Using recreational angler participation data during an 11-year study period from 73 public waterbodies in Nebraska, USA, we developed a resource size-use model that demonstrates the ability of natural resource system size to indicate the quantity of recreational use they receive. We demonstrate how resource size-use models can estimate use for unsampled systems, produce broad-scale estimations of use, guide the allocation of resources, and predict how changes in resource system size may affect use. Resource size-use models provide opportunities to manage recreational use, which has been previously elusive for social-ecological systems.

1. Introduction

Size-based metrics, such as population size or corporate firm size, often dominate decisions and policy in social systems. For example, the amount of federal funding a town or city receives in the USA is based on population size. Yet, size-based metrics have not been fully appreciated or adopted in the management of coupled social and ecological systems. A size-based metric could assist with quantifying resource use within complex social-ecological systems and improve management. Resource use is often related to public and political support, ecosystem services, social conflicts, and ecological disturbances (e.g., Thomas and Reed 2019; Arlinghaus et al., 2020; DaRugna et al., 2022). For example, congressional acts, such as the Wild and Scenic Rivers Act and the National Trails Act were passed due to the close ties of public land protection and outdoor recreational use (Clawson and Knetsch, 1996; Thomas, 2009; Thomas and Reed, 2019). Increases in outdoor recreational use can also lead to elevated environmental impacts (Monz et al., 2013; Jedd et al., 2018; DaRugna et al., 2022). To this end, managing outdoor recreational use is paramount for retaining the many benefits and key ecosystems services that natural resource systems (a specified, designated managed area containing forested areas, wildlife, or water systems, such as a reservoir, mountain, or wildlife refuge; Ostrom, 2009) provide.

Estimating and monitoring resource use in social-ecological systems is logistically difficult and costly (e.g., Post et al., 2002; Hadwen et al., 2007; Trudeau et al., 2021). The spatial distribution and composition of natural resources across the landscape contributes to the difficulty of quantifying use across multiple resource systems (e.g., Carpenter and Brock, 2004; Parry et al., 2009; Wilson et al., 2016). Not all resource systems receive the same amount of use (e.g., Steffe et al., 2008; Askey et al., 2018; DaRugna et al., 2022). The users of these resource systems also contribute to the variation in use, as users represent diverse and heterogeneous groups (e.g., Holland and Ditton, 1992; Connelly et al., 2001; Watkins et al., 2018). For example, recreational anglers are geographically diffuse, diverse in their motivations, and behaviorally dynamic (e.g., Arlinghaus, 2006; Golden et al., 2019; Kane et al., 2020). Similarly, hunters are heterogeneous in where they hunt, how frequently they hunt, and their motivations to hunt (e.g., Hunt et al.,

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2005; Kerr and Abell, 2016; Hinrichs et al., 2021). Despite the importance of recreational use, the difficulties in quantifying use precludes effective management – which is a problem that could be remedied with the ability to predict recreational use.

Studies have aimed to predict recreational use patterns using a wide variety of social and ecological variables (e.g., Hunt 2005; Post et al., 2008; Johnston et al., 2010). Many have highlighted that resource system size (e.g., surface area) can reliably predict resource use (e.g., Lyach and Čech, 2018; Kaemingk et al., 2019; Trudeau et al., 2021). None of these studies, however, attempted to predict resource use using solely resource system size. Size of the resource is an important metric that influences ecological and social aspects of natural resources, and thus, may serve as an important indicator for predicting resource use. For example, the size of floodplain waterbodies, along with depth and water clarity, is important for structuring fish assemblages (Miranda and Lucas, 2004; Lubinski et al., 2008; Miranda, 2011). Larger waterbodies typically have greater species richness for a variety of taxa and can offer more diverse recreational opportunities compared to smaller waterbodies (e.g., Post et al., 2000; Hunt, 2005; Nikolaus et al., 2021). Similarly, land area determines the habitat management and conservation costs (Armsworth et al., 2011), and larger land fragments often produce more ecosystem services (Hartter, 2010). Size can also be related to systems thinking and resilience properties such that larger resource systems may also function as buffers and can withstand increased amounts of recreational use and disturbance (Gunderson and Pritchard, 2012). Furthermore, resource size may serve as a composite variable in that it represents several coupled social-ecological relationships that are strongly correlated with resource size. Our previous work identified that fish stocking patterns and angler use were best explained by waterbody size; small waterbodies received more fish stocked and angler effort (Kaemingk et al. 2022). We therefore posit that the size of a resource system could be used as a proxy to infer resource use, based on the empirical relationships between resource size and resource use. Therefore, the development of resource system size-use models could provide the groundwork for managers to better optimize the social and ecological benefits of these systems.

Using resource system size as an indicator of recreational use is attractive because it is 1) cost-effective compared to traditional onsite surveys, 2) readily available or can be quantified using GIS and remote sensing techniques (Pekel et al., 2016), and 3) likely to be widely adopted by managers because of the aforementioned properties and simplicity. Resource size-use models can produce broad-scale estimations of use, providing a baseline for management by enabling natural resource managers to predict the amount of use at all resource systems within their management region. Another utility of resource size-use models is guiding the allocation of management resources according to expected use. Resource size-use models can identify deviations from expected resource system use, highlighting priority systems in the allocation of limited resources. Finally, natural resource managers can also use developed resource size-use models to predict changes in resource use caused by changes in resource size, such as forecasting whether anthropogenic changes, like climate change, will ultimately influence the quantity of resource use based on changes in the size of natural resource systems (e.g., drought and deluge periods).

Our goal is to develop a simple resource size-use model that can be used to improve the management of social-ecological systems, using a large recreational fishery dataset with information on angler use of waterbodies from Nebraska, USA. We then illustrate the utility of this model by applying it to three fisheries management case scenarios that allow: 1) Estimation of angler use on unsampled waterbodies across multiple spatial scales; 2) Identification of under- or over-used waterbodies; 3) Prediction of how angler use would respond after a change in waterbody size. Our hope is to demonstrate that the benefits of using size-based metrics go beyond social systems by also improving our understanding of social-ecological systems (Kaemingk et al., 2019; Kaemingk et al. 2022). Resource size-use models could provide natural resource managers the opportunity to predict natural resource use and leverage these predictions to influence all aspects of natural resource management. With resource size-use models, managers are equipped with a necessary tool to effectively manage resource use, a critical shortcoming of most social-ecological systems management.

2. Methods

2.1. Study area

We assessed recreational angler use (i.e., angler effort in hours) at 73 public waterbodies throughout Nebraska, USA from 2009 through 2019 (Table S1 in Appendix S1), which ranged in size (i.e., surface area) from 1 to 12,141 ha (mean = 593 ha; standard deviation = 2,028 ha). The waterbodies were reservoirs constructed for a variety of purposes including flood control, irrigation storage, hydropower generation, and community recreation. These waterbodies were spatially distributed throughout Nebraska and represented a diversity of fishing opportunities (Pope et al., 2016; Kaemingk et al., 2020).

2.2. Angler use estimations

We obtained estimations of angler use (hours spent fishing) from instantaneous counts of bank anglers and angling boats at each waterbody. Counts occurred between sunrise and sunset from April through October. Angler-count days and times were randomly selected following a stratified multi-stage probability-sampling regime (Malvestuto, 1996). Angler-use estimations were calculated using previously described methods (Malvestuto et al., 1978; Pierce and Bindman, 1994; Pollock et al., 1994; Malvestuto, 1996; Pollock et al., 1997). We conducted angler counts for 10 to 24 days per month, depending on the size of the waterbody and logistics (Kaemingk et al., 2018). During each month, angler counts were stratified by day type (i.e., weekdays and weekend days, holidays were either treated as weekend days or their own day type) and day periods (i.e., morning and afternoon). The number of anglers counted was multiplied by the number of hours in each survey period and divided by the probability of selecting a day period (0.5) to produce a daily use estimation, which was multiplied by the number of days within a day type present in the month and summed across all day types to produce a monthly angler use estimation. Monthly angler-use estimations were summed to estimate angler use from April through October, from here on referred to as annual angler use. For waterbodies that were sampled multiple years, annual angler use was averaged across all years sampled.

2.3. Analysis

We used linear regression, one of the simplest methods of linking explanatory and response variables (Milton et al., 2019), to test for an expected resource system size-use relationship between annual angler use and waterbody surface area. We then used the coefficient of determination (r^2) and corresponding p-value to evaluate the strength and determine significance ($\alpha = 0.05$) of the resource system size-use relationship. Waterbody surface area was determined from the surface area at conservation pool. We $\log_e$-transformed annual angler use and waterbody size to reduce heteroscedasticity and represent the expected diminishing effect of increasing waterbody size on annual angler use (Parsons and Kealy, 1992; Woolnough et al., 2009; Hunt and Dyck, 2011). When predicting angler effort based on our resource system size-use relationship, we used the predicted value from our model and the associated 95 % confidence interval. We conducted all analyses in R (R Core Team, 2017).

3. Results and discussion

Waterbodies included in our assessment varied in annual angler use,

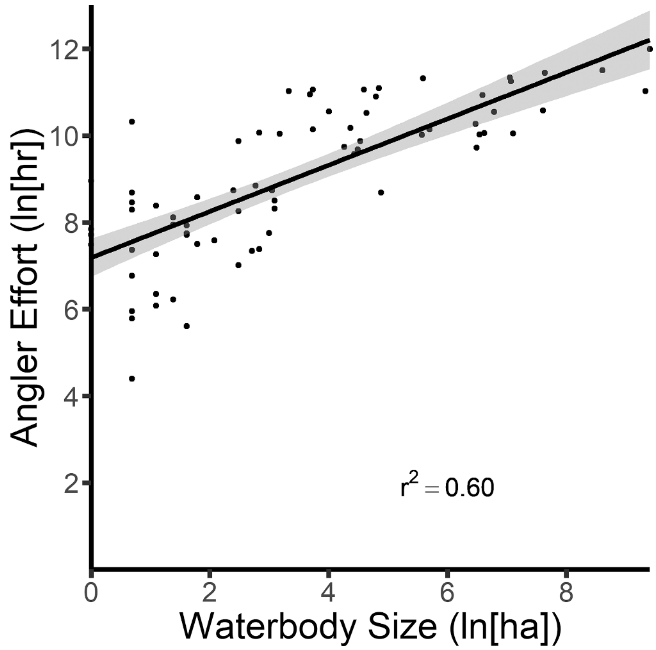
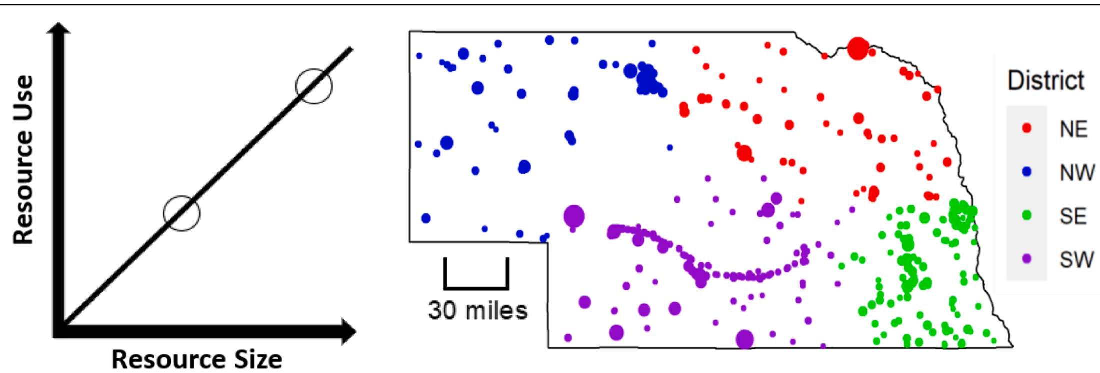


Fig. 1. Model displaying the relationship ($r^2 = 0.60$) between annual angler use (natural log of the annual extrapolated use in hours) and waterbody size (natural log of hectares). Points represent individual waterbodies sampled either once or across multiple years, and the ribbon represents a 95 % confidence interval.

ranging from 81 h to 161,774 h (mean = 23,560 h; standard deviation = 30,793 h). Linear regression analysis revealed that waterbody size was a significant predictor of annual angler use ($r^2 = 0.60$, $p < 0.01$; Fig. 1). Angler use was positively correlated with waterbody size ($\ln[\text{use}] = 7.1861 + 0.5338 \times \ln[\text{size}]$). The y-intercept of 7.19 indicated that each 1-ha waterbody in Nebraska received about 1,330 h of annual angler use.

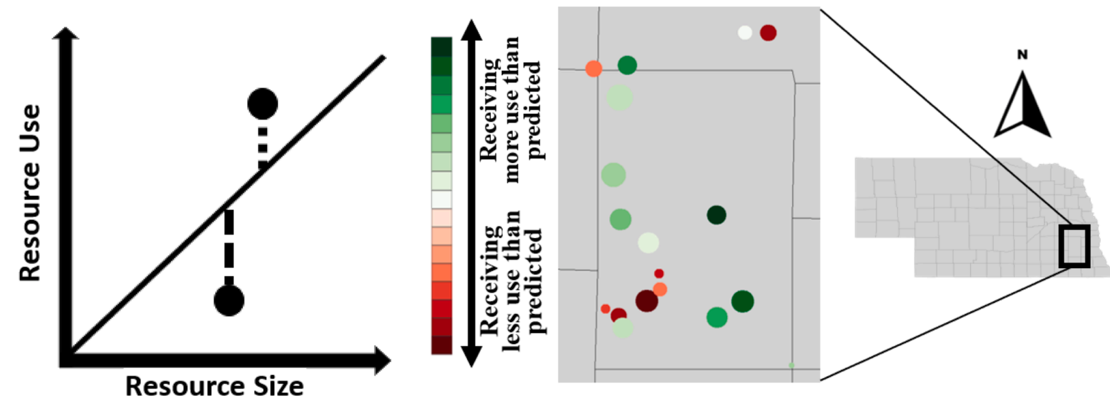
Resource size-use models have many potential management benefits. We anticipate that estimating resource use from random subsamples across the entire range of resource system sizes can be used to develop these resource size-use models. For example, we measured resource use at 73 out of the 646 public waterbodies in Nebraska to build our resource size-use model. The waterbodies we sampled ranged from 1 to 12,141 ha in size, vary in their location on the landscape (e.g., vary in their distance from population centers or other waterbodies), and each represent unique angling opportunities. Thus, these waterbodies were representative of all public Nebraska waterbodies. We expect that resource size-use models will be stable for several years, but should be re-calibrated if major changes in population distribution or availability of resource systems on the landscape occurs (Hunt et al., 2019a; Hunt et al., 2019b; Kaemingk et al., 2021). We anticipate that the development of resource size-use models for different regions, resource user groups, and across different spatial (e.g., local, regional, national) and temporal (e.g., seasonal, annual, decadal) scales will improve the management of other social-ecological systems, beyond recreational fisheries. Further research is needed to determine how many natural resource systems should be sampled to build an appropriate model and whether the strong resource size-use relationship for Nebraska holds in other regions with higher or lower availability of resources.

Whilst we expected a significant relationship between angler use and waterbody size, the amount of variation explained in angler use by waterbody size is somewhat surprising. Nebraska has a unique social-ecological landscape, with much of its population residing in the eastern half of the state and most of the larger waterbodies in the western half of the state. This mismatch in urban proximity to large resources (that are expected to receive greater use) could have limited our ability to develop a resource size-use model, given that distance and associated travel costs often predict recreational use (Berman and Kofinas, 2004; Hunt et al., 2011; Wilson et al., 2020). Similarly, there are likely heterogenous socio-economic conditions across the state that could have introduced a large amount of variation in angling use to further weaken this relationship (Searle and Jackson, 1985; Shores et al., 2007). Additionally, the values (data) used to develop the resource size-use model contained uncertainty (i.e., not measured without error or variance) in both x- and y-axes. However, we demonstrate that this approach is robust to Nebraska's social-ecological landscape and data uncertainty, thus warranting testing in other areas. Future development of other resource size-use models will determine how robust this



Box 1. Estimation of use on unsampled resource systems across multiple spatial scales.

Resource size-use models can provide resource use estimation for unsampled waterbodies across designated management units. Nebraska's public lakes and reservoirs are divided into four fishery management districts. In the map above, each circle represents a publicly managed waterbody, its size represents the amount of angling effort it is predicted to receive, and its color represents the corresponding management district. Based on the size-use relationship, annual angler use in the 4 districts ranged from 852,090 (95 % CI: 598,703 – 1,213,920) hours (SE) to 1,506,952 (95 % CI: 1,039,618 – 2,186,471) hours (NW) per district (mean = 1,187,638 h per district; standard deviation = 331,212 h). Effort estimates from each district can be summed to predict that there are about 4,750,551 (95 % CI: 3,271,489–6,911,228) hours of annual angler use on Nebraska's public waterbodies (excluding streams and rivers). The aforementioned 95 % CI's represent the models residual uncertainty.



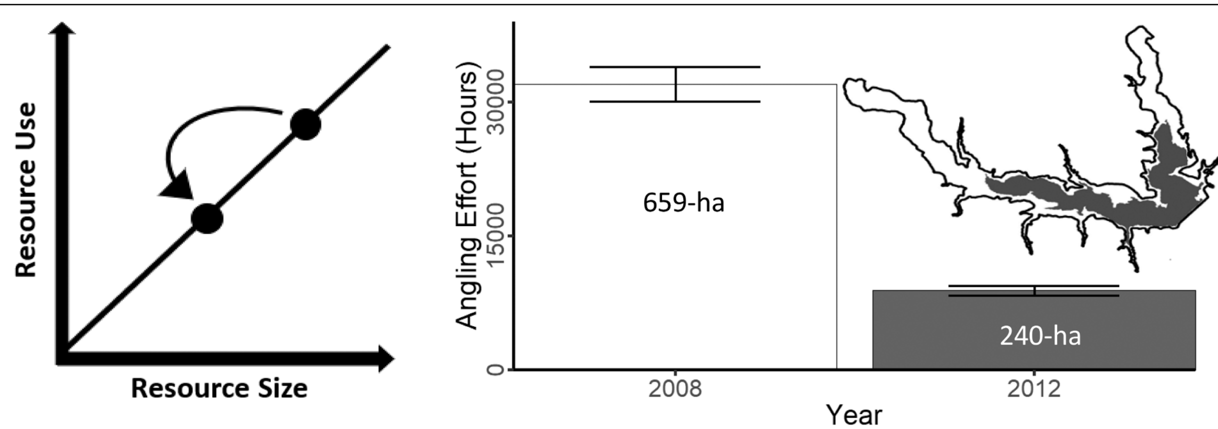
Box 2. Identification of under- or over-used resource systems.

Resource size-use models can provide information about expected angler use, such as whether waterbodies are receiving more or less use based on size. Natural resource managers can compare predicted levels of use with measured levels of use to highlight resource systems that deviate from the predictions based on their size (positive or negative residuals). The above map represents publicly managed waterbodies in and around Lancaster County, Nebraska, circle size represents the amount of angling effort each waterbody is predicted to receive, and its color indicates the amount of angling use the waterbody receives compared to the amount predicted by its size. The most under-used waterbody received 9,842 h of use less than predicted and the most over-used waterbody received 58,612 h of use more than predicted, based on the developed resource size-use model. Natural resource managers can then use this information to help determine where to allocate specific management resources.

approach is.

The resource system size-use model provides an easy and cost-effective method of obtaining broad-scale use estimations. Natural resource managers can estimate use for all the resource systems within their management region, including systems that have not been sampled (Box 1).

Estimates of resource use can provide meaningful baseline information about how much use an average system of a specific size should receive. Natural resource managers could produce statewide, nationwide, and ultimately worldwide estimations of resource use through existing social-ecological datasets (Lynch et al., 2021) and remote sensing techniques (Pekel et al., 2016). These broad estimations of resource use can provide utility in the extrapolation of benefits that natural resource systems provide, such as economic benefits (e.g., Bergstrom et al., 1990; Lazarow, 2007; Spirk et al., 2008). Caution must be taken, however, to not estimate outside the bounds of the model. Our



Box 3. Prediction of use after a change in resource system size.

Resource size-use models can provide predictions of how angler use would respond after a change in waterbody size. In 2009, a Frontier County, Nebraska reservoir decreased in size from 659 ha (represented by the white area; right panel) to 240 ha (represented by the black area; right panel) to allow for dam repair (Chizinski et al., 2014). We used our resource system size-use model to attempt to predict how use could have changed with the reduction in resource system size (left panel). In this case, we predict that use of the 659-ha waterbody would drop from 42,222 (95 % CI: 28,275–63,076) hours of annual angler use to 24,600 (95 % CI: 16,759–36,206) hours of annual angler use if the waterbody remained at 240 ha in size. Indeed, Chizinski et al. (2014) documented a decrease in annual angler use in the years following the drawdown (right panel). Although the actual use was less than predicted by the model both pre- and post-drawdown, the model provides insight into how the quantity of recreational use may change as a result of a change in resource size.

model was constructed to predict resource use among recreational fisheries in Nebraska, thus, it may not be appropriate to use elsewhere. Instead, the development of additional resource size-use models for different types of resource use and in different areas is needed.

The resource system size-use relationship also provides utility in the prioritization and allocation of natural resource management funds. Identifying resource systems that receive less use than predicted by size, for instance, may provide managers with insight about specific mechanisms that deter recreational use. Additionally, if managers identify a resource system that is receiving less use than predicted by size, they may decide to invest more resources in that system to increase use. For example, angler use typically increases after a fish stocking event (e.g., Loomis and Fix, 1998; Baer et al., 2007). Fish stockings could be directed at resource systems that are receiving less use than predicted based on their size. Alternatively, when funding is limited, managers may utilize the resource system size-use relationship to help determine which waterbodies may receive a decrease in funding. At the landscape-scale, natural resource managers can compare regional resource systems in terms of their predicted (Box 1) and measured levels of use and utilize this information to guide the allocation of resources across multiple resource systems (Box 2). On one hand, large deviations from predicted use, for systems where the quantity of use has been sampled, may represent the potential occurrence of social conflicts or ecological disturbances, such as overcrowding and declines in native biodiversity (e.g., Cole, 2001; Dudgeon et al., 2006; Thompson, 2015). Further investigations into these systems may provide insights into additional metrics that may improve future, more complex models. Developing more complicated models, however, may limit their utility among natural resource managers.

Another benefit of creating resource system size-use models is the ability to predict how resource system use might change if the size of a resource system were to change (Box 3). For instance, water may be drained from a reservoir to manage fish populations or to repair physical structures of a waterbody (e.g., Chizinski et al., 2014). Future changes to the size of waterbodies and other natural resource systems may occur because of anthropogenic influences such as climate change, irrigation, and wildfires; (e.g., Gao et al., 2011; Bawa, 2017; Zou et al., 2017). However, there are additional reasons why recreationalists may be attracted to, or repelled from, resource systems that have undergone a change in size. For instance, the draining of a reservoir may condense fish populations, attracting catch-focused anglers. That same management action may result in a muddy shoreline, repelling away those wanting to use the reservoir for aesthetic purposes (e.g., Moeller and Engelken, 1972; Hunt, 2005; Hunt et al., 2019a; Hunt et al., 2019b). To this end, caution is required when using a simple model to make predictions on dynamic systems. Consideration must be given to both the type of resource system and how changes in its size might affect the specific group of users being considered.

Developing resource system size-use models can change how our natural resources are managed by providing broad-scale estimations of resource system use, guiding the allocation of management resources according to expected use, and revealing how different user groups interact with natural resources. Resource system size-use models will allow natural resource managers the opportunity to quantify, predict, and manage use across all resource systems, enabling a landscape approach to resource use management (Matsumura et al., 2019; van Poorten and Camp, 2019) that was previously costly and logistically difficult. Building resource size-use models, at the appropriate scales, provides natural resource managers with a tool to improve the management of resource use in complex social-ecological systems.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data used are provided in the Appendix.

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Authors' contributions

D. S. Kane, M. A. Kaemingk, and K. L. Pope conceived the initial idea. K. L. Pope and C. J. Chizinski secured funding for the project. K. D. Koupal and M. A Pegg provided input to improve the study design. M. A. Kaemingk, C. J. Chizinski, and K. L. Pope provided oversight of data collection. D. S. Kane analyzed the data and wrote the initial draft of the manuscript. All authors contributed critically by editing drafts of the manuscript and gave final approval for publication.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2022.109711.

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