The Ecological, Economic and Social Dimensions of Marine Plastic Pollution and Marine Plastic Pollution Interventions

by

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ABSTRACT

Marine plastic pollution (MPP) has emerged as one of the most pressing global environmental challenges of the anthropocene. There has been an upsurge in investment to mitigate MPP; however, interventions can be costly, inequitable, and ineffective in achieving their objectives. In my dissertation, I aim to research key considerations for creating cost-effective, equitable mitigation strategies for MPP and its impacts to marine biodiversity and coastal communities. In chapter one, I introduce the challenges plastic pollution poses. In chapter two, I use seascape ecology theory to present the concept of the plastic-scape and describe how seascape ecology principles, methods, and approaches to transdisciplinary science can inform research to mitigate MPP. In chapter three, I present a framework to help decision makers estimate the total cost of MPP interventions and partial costs accrued by stakeholder groups. I then apply this framework to two quantitative case studies and four comparative case studies to exemplify its use and highlight the ways spatial scale, temporal scale, and socio-economic conditions influence the intervention cost and cost distribution. In chapter four, I employ a trait-based approach to produce a framework for developing indices of species vulnerability to macroplastic pollution. Finally, in chapter five, I implement the framework developed in the previous chapter and present a multi-taxonomic, macroplastic vulnerability index for three marine taxa—mammals, birds, and turtles—to identify the marine species most vulnerable to macroplastic pollution in Hawaiʻi. Overall, my dissertation shows how policy-driven, systemic research of MPP and its interventions can improve efforts to address MPP and its socio-economic and ecological consequences.

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CHAPTER 1

INTRODUCTION

Over the last few decades, there has been exponential growth in plastic production, with a projected increase to 33 billion tons annually by 2050 (Rochman et al. 2013). Our capacity to manage this material has not matched increases in production. As a result, plastic pollution has emerged as one of the most pressing sustainability challenges of our time, occurring in every marine ecosystem across the globe (Eriksen et al. 2014). In 2016, between 19.3 and 23.4 million metric tons of plastic entered aquatic ecosystems, and this number will continue to increase despite commitments to address it (Borrelle et al. 2020). This pollution is having detrimental effects on the environment, the economy, and human wellbeing across the globe (Santos et al. 2021; Bucci et al. 2020; Beaumont et al. 2019; Markic et al. 2018; Brouwer et al. 2017; Newman et al. 2015; Mouat et al. 2010).

Interactions with marine debris and plastics have been documented in more than 1300 marine species, including every species of sea turtle, 70% of marine mammals and 55% of seabirds (Santos et al. 2021; Kühn and Van Franeker, 2020). The community and ecosystem level impacts of plastic pollution remain understudied, but the extent of individual interactions documented suggests plastic may shift marine ecosystems (Bucci et al. 2020). Furthermore, of the few studies evaluating impacts at the community and ecosystem level, several have shown detrimental outcomes. For instance, fishing gear may reduce the health of coral reef ecosystems (Donohue et al., 2001; Lewis et al. 2009), and plastic interaction increases coral disease prevalence (Lamb et al. 2018).

Marine plastic pollution has global economic consequences, with coastal communities bearing the greatest burden (Mouat et al. 2010; Newman et al. 2015). These impacts include (1) increased expenditures due to marine plastics, (2) the diminished value of marine natural capital and (3) increased healthcare costs (Newman et al. 2015).

Direct expenditures due to plastic pollution are significant. For example, marine plastics cost the 21 economies of the Asia-Pacific rim $1.26 billion dollars (2008 USD) annually (McIlgorm et al. 2011). These costs fall primarily on coastal municipalities and marine sectors including tourism, coastal agriculture, fisheries and aquaculture, shipping, and emergency rescue services (Newman et al. 2015). These costs have been most well documented for clean-up efforts and the fisheries sector. The total cost of clean-ups for coastal municipalities in the UK are between €17,936,000 - €18,780,000 each year (Mouat et al. 2010). The city of Long Beach, California spent $1,837,390 in one year on beach clean-ups alone (Leggett et al. 2014). On average, Selayar fisherman spend roughly 350 rupiahs per year on gear and vessel repairs due to interactions with plastic (Hermawan et al. 2017), and interactions with plastic reduce the revenue of the Scottish fishing industry by 5% annually (Mouat et al. 2010). Vessel damage and downtime can also affect recreational boating, shipping, and rescue services, but underreporting makes it difficult to estimate total costs to these industries (Mouat et al. 2010).

Beaumont et al. (2019) estimate the value of marine ecosystem services decrease between $3,300 and $33,000 per ton of plastic pollution annually, a 1-5% reduction in marine natural capital (based on 2011 values for ecosystem services and marine litter). However, this estimate does not include the direct loss of ecosystem services from plastics in the marine environment, such as the reduction in the aesthetic, religious, or recreational values of the ocean. For example, the presence of plastic pollution on beaches can reduce the recreational value of beaches by tens of millions of dollars a season (Stickel et al. 2012, Jang et al. 2014), suggesting the total value of ecosystem services lost is even greater than the Beaumont estimate.

Marine plastic pollution increases healthcare costs via direct interaction with plastics and through chemical contamination. Direct interactions can occur while boating, diving or on the beach. In New Zealand, 82,891 active beach injury claims were accepted over a 10-year period. The average claim was NZ$450, totaling NZ$37,301,000 (Campbell et al. 2019). Plastic debris has also been found in fishes and bivalves sold for consumption (Rochman et al. 2015). The impacts of plastic-derived chemicals on human health are not yet well-documented, but many of the polymers used in the development of plastics have been recognized as hazardous or carcinogenic (Lithner et al. 2011).

Plastic pollution also has social impacts. Litter can undermine the psychological benefits of the coastal environmental (Wyles et al. 2016). Studies conducted around the world indicate that beachgoers place a high value on clean beaches and are willing to pay or participate in volunteer clean-ups to maintain them (Schuhmann et al. 2016; Brouwer et al. 2017; Shen et al. 2019). MPP’s social and economic impacts disproportionately affect marginalized communities, perpetuating environmental injustices (Phelan et al. 2020). Marine ecosystems are critical to the livelihoods and cultural wellbeing of low-income coastal communities and the impacts of marine plastics disproportionately affect these groups (Lau et al. 2019).

Despite extensive research into the extent and effects of plastic pollution, there is still limited knowledge on how to effectively reduce plastic pollution and its effects, in an equitable and cost-effective manner. This dissertation aims to answer policy-driven research questions that explore the gap between our understanding of the environmental, economic, and social impacts of plastic pollution and how this should inform effective management.

This dissertation is separated into four main chapters. In chapter two, I present the concept of the plastic-scape and explore how seascape ecology principles, methods, and approaches to transdisciplinary science can inform research to mitigate marine plastic pollution. In chapter three, I explore the following research questions: What costs should be considered in the net cost of marine plastic pollution interventions; how should costs be categorized; and how do spatial scale, temporal scale, and socio-economic context influence intervention cost? I present a framework to estimate the net cost of MPP interventions and partial costs accrued by stakeholder groups. I then apply this framework to two quantitative and four comparative case studies to exemplify its use and highlight the ways spatial scale, temporal scale, and socio-economic conditions influence the intervention cost and cost distribution. In chapter four, I explore what traits influence population level vulnerability to ingestion and entanglement of macroplastics. I then employ a trait-based approach to develop a framework for creating vulnerability indices of the physical impacts of macroplastic pollution. Finally, in chapter five, I implement the framework presented in chapter four to answer the following research questions: What is the relative vulnerability of three marine taxa in Hawaii—mammals, birds, and turtles—to macroplastic pollution, and which species are most vulnerable? My results represent the first multi-taxonomic vulnerability index for macroplastic ingestion and entanglement.

CHAPTER 2

THE PLASTIC-SCAPE: APPLYING SEASCAPE ECOLOGY TO MARINE PLASTIC POLLUTION

Introduction

Marine plastic pollution (MPP) is an urgent sustainability challenge. In 2016 alone, between 19.3 and 23.4 million metric tons of plastic entered aquatic ecosystems (Borrelle et al. 2020). This pollution has environmental, economic, and social consequences (Beaumont et al. 2019), which have inspired global stakeholder action (Schnurr et al. 2018; Xanthos and Walker 2017). Still, even if these ambitious efforts are achieved, plastic pollution emissions will continue to rise due to increased production (Borrelle et al. 2020). As MPP continues to increase, so will its social, ecological, and economic consequences (Beaumont et al. 2019).

Current management efforts for MPP are often ad hoc, without consideration for decision-makers' goals, scale of governance, context of implementation, or systematic coordination across scales and sectors (Excell et al. 2018). Intervention efficacy is rarely evaluated and evaluated interventions report mixed outcomes (Excell et al. 2018). For example, bag regulations are among the most popular policies for plastics across the globe, yet less than half have been evaluated for effectiveness in reducing bag consumption, and 40% of evaluated policies have achieved little to no impact (Excell et al. 2018). In general, the effectiveness of top interventions—bag bans and levies, deposit refund schemes, and dumping fines—are conditional on the context of implementation, including governance, socio-economic status, and environmental conditions (Excell et al. 2018; Lavee 2010; Oosterhuis et al. 2014; McIlgorm et al. 2011).

Effectively implemented policies may still fail to reduce MPP. Research has shown that even if the most ambitious global commitments are achieved, annual plastic emissions will continue to increase due to increased production driven by global development and population growth (Borrelle et al. 2020). This indicates that the suite of solutions being implemented are largely insufficient for addressing the primary sources and environmental pathways of MPP.

Finally, effective policies must ultimately reduce the social and ecological consequences of MPP, which depend on how MPP interacts with social and ecological communities. Not all ecosystems are equally vulnerable to MPP, and marine regions vary in their importance to human communities (Murphy et al. *in review;* Armoškaitė et al. 2020; Beaumont et al. 2019). As a result, policy effectiveness should not only be measured by MPP reduction, but also by social-ecological outcomes.

Failure to mitigate MPP and its consequences through current efforts has fueled calls for transformative, system-wide change along the entire plastics' life cycle (Borrelle et al., 2020; Raubenheimer and Uhro, 2020). This will require action across scales of governance that not only consider policy objectives, but also feasibility, cost, trade-offs, and efficacy for mitigating the social, ecological, and economic consequences of MPP (Tessnow-von Wysocki and Le Billon 2019; Murphy et al. 2021; Helm et al. 2022). This approach must 1) be transdisciplinary, 2) be multi-scale, 3) be spatially-explicit, and 4) encompass the entire *plastic-scape*—which includes all the governance systems, human actors, and ecological components (i.e., abiotic, and biotic processes) that contribute to patterns of plastic production, use, and pollution, as well as the interactions between MPP and human and natural communities that drive its social and ecological consequences (Figure 1).

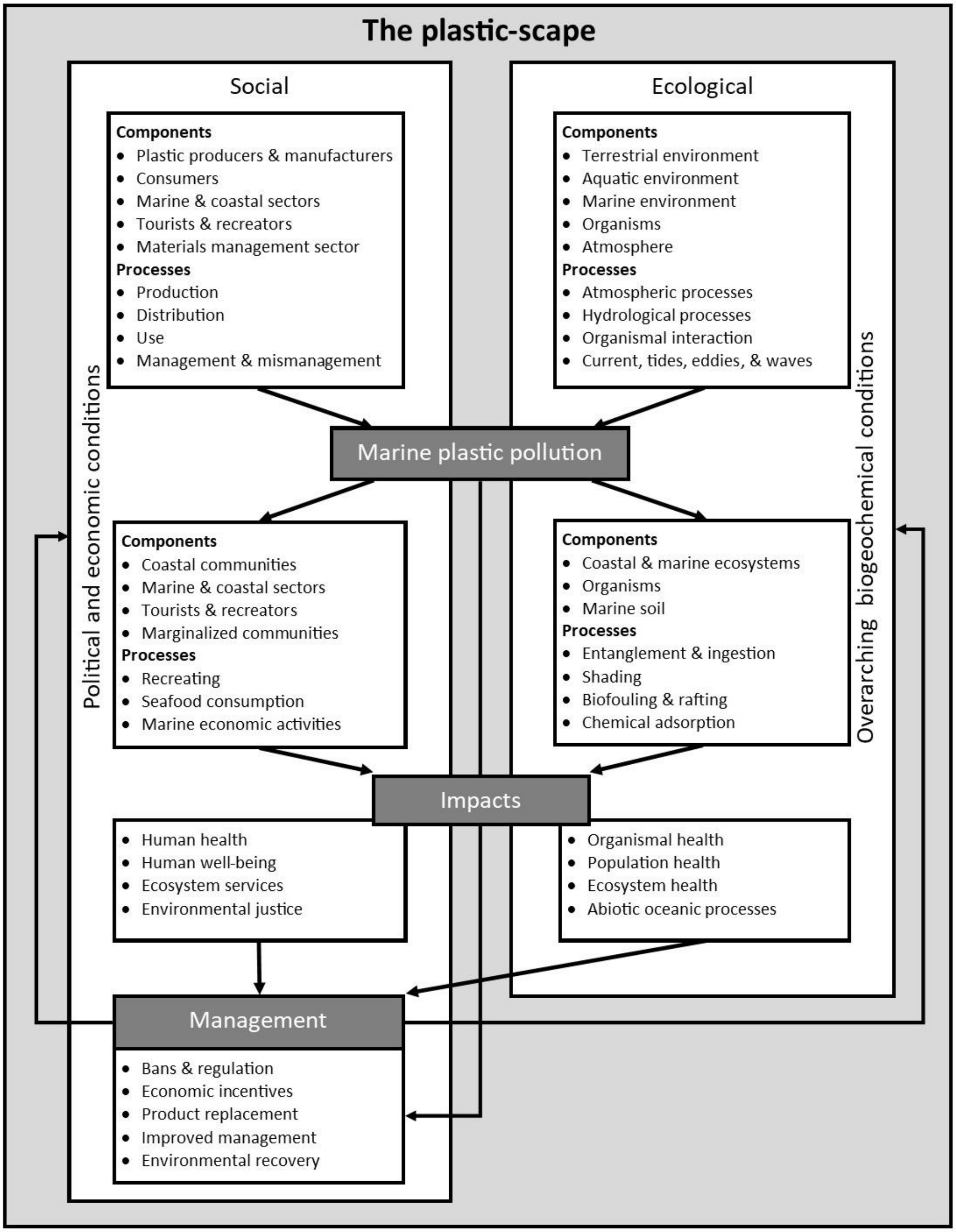

Figure 1 Conceptual model of the plastic-scape. The first set of social-and ecological components and processes drive the creation and distribution of marine plastic pollution. The second interact with marine plastic pollution to drive the social-and ecological impacts of marine plastic pollution. Finally, marine

plastic pollution and its impacts drive management actions that can act along the entire plastic-scape. The social and ecological components of the plastic-scape also interact with and influence each other.

Landscape ecology (LE) provides a spatially explicit, multi-scale approach for understanding social-ecological landscapes that is well-suited for MPP research and management (Opdam et al. 2018; Wu 2013). LE draws on natural and human ecology, geography, history, economics, and wildlife management to understand the relationship between pattern and process in the environment (Risser et al. 1984; Wu 2013). Historically, European LE focused on human landscapes and solutions-oriented questions, while North American LE aimed to advance quantitative methods for understanding natural systems (Wu and Hobbs 2002). The integration of these approaches provides theory, principles, methods, and tools for studying complex and spatially explicit environmental challenges (Wu 2013). Additionally, LE's contributions to sustainability science, environmental management, and conservation demonstrate its value in achieving conservation outcomes (Opdam et al. 2018; Wu 2006).

More recently, seascape ecology (SE) has emerged (Pittman 2018). Like LE, it is well-suited to support sustainability science and has informed several marine conservation issues (e.g., habitat restoration, marine planning), but its application to MPP has been limited (Rees et al., 2018; Stamoulis and Friedlander 2013; Fraschetti et al. 2009).

SE offers a multi-scale approach for understanding and evaluating the *plastic-scape* (Cumming et al. 2017; Opdam et al. 2018). Below, I explore opportunities for applying SE to MPP research and management.

The Seascape Ecology Approach

A seascape ecology approach can help address the shortcomings of the current approach by providing a framework that 1) is spatially explicit, to account for context of implementation, 2) is holistic and multi-scale, to ensure that the sum of individual interventions is enough to address this global challenge, and 3) integrates social and ecological outcomes.

The maturation of SE has promoted the emergence of seascape specific principles, tools, and methods to capture the dynamic and three-dimensional structure of the seascape, which is necessary for understanding MPP (Kavanaugh et al. 2016; Lepczyk et al. 2021; Swanborn et al. 2022; Wedding et al. 2011). It has also sparked interest in novel research priorities—seascape connectivity; seascape goods and services; ecosystem-based management; and applications for marine management (Pittman et al. 2021). This has driven novel approaches for evaluating these seascape components, which are important aspects of the *plastic-scape* that have been difficult to quantify (Urlich et al. 2022; Barbier and Lee 2014; Halpern et al. 2010; Grober-Dunsmore et al. 2009).

Landscape sustainability science, another emerging subdiscipline, aims to understand how landscape structure and elements influence the sustainability of real-world landscapes, including biodiversity, ecological processes, ecosystem services, and human wellbeing (Wu 2021). To center human dimensions of the landscape, the landscape sustainability science framework captures a broader set of landscape pattern drivers than traditional LE—socioeconomic, political, technological, natural, and cultural—all of which are important in the *plastic-scape* (Bürgi et al. 2005). Further, landscape sustainability science is inherently transdisciplinary and applied. Therefore, approaches from this field can be used to inform transdisciplinary research and management approaches for the *plastic-scape* (Wu 2021).

Below, I describe the ways SE principles can inform our understanding of the *plastic-scape*, describe applicable methods and tools for evaluating the *plastic-scape*, and discuss how LE and SE transdisciplinary research approaches can improve research and management.

Concepts from seascape ecology

Heterogeneity and pattern-process relationships. Heterogeneity is the spatial variation—or patterns—in a seascape, represented as patches or gradients (Pittman 2018; Wu 2012). Composition relates to the number and proportion of patch types, while configuration relates to their spatial arrangement (Gustafson 1998).

The *plastic-scape* is heterogenous in both its social and ecological dimensions. Patterns in MPP configuration exist, such as gradients throughout the water column and high-density patches in the gyres and coastal zones (Brignac et al. 2019; Eriksen et al. 2014; Hardesty et al. 2017). These patterns are well-represented in the MPP literature; however, the social-ecological components of the plastic-scape also have patterns, making the impacts of MPP on biodiversity, human health, marine ecosystem services, and human well-being heterogenous (Barbier and Lee 2014; Bucci et al. 2020; Phelan et al. 2020). Heterogeneity in these other dimensions must also be considered to effectively address MPP and its consequences more broadly.

Processes are dynamic features that create and are influenced by seascape patterns (Boström et al. 2011; Fu et al. 2011; Turner 1989). Seascape connectivity—the movement of living and non-living material from one location to another—is an important component of these pattern-process relationships (Hyndes et al. 2014; Olds et al. 2018; Olds et al. 2016). Most MPP is derived from land-based sources, which makes understanding land-sea connectivity and connectivity between human-dominated and natural ecosystems critical (Napper and Thompson 2020).

Processes influencing the *plastic-scape* represent all five drivers from landscape sustainability science—socioeconomic, political, technological, natural, and cultural (Bürgi et al., 2005). Socioeconomic, technological, cultural, and political processes affect the patterns of plastic production, use, management, and mismanagement in our environment, ultimately shaping the pathways of plastic leakage (Thushari and Senevirathna 2020; Napper and Thompson 2020). They also influence patterns of plastic type, shape, and chemical composition in the ocean (Thushari and Senevirathna 2020; Napper and Thompson 2020). The human processes driving patterns in the plastic-scape are influenced by the overarching geopolitical and socio-economic context, such as patterns of human population density, wealth, and governance (Borrelle et al. 2020; Jambeck et al. 2015).

Natural processes also drive patterns in the *plastic-scape*. Ecological processes (e.g., rainfall, animal movements) influence patterns of plastic leakage from management sites, such as landfills (Axelsson and van Sebille 2017; Ballejo et al. 2021). Once in the environment, hydrological processes are one of the primary pathways for transporting terrestrial plastic pollution to the ocean, making watershed patterns important for informing patterns of MPP (Correa-Araneda et al. 2022; Windsor et al. 2019, Lebreton et al. 2017).

Oceanographic processes—currents, tides, and eddies—are the primary processes driving MPP transportation and deposition in the ocean (Eriksen et al. 2014, Brignac et al. 2019). Interactions with animals (e.g., ingestion), plants (e.g., entanglement), bacteria (e.g., biofouling), and human activities (e.g., clean-ups) also contribute (Jacquin et al. 2019; Kaiser et al. 2017; Ocean Conservancy 2016; Ryan 2020; Sanchez-Vidal et al. 2021). Understanding the relationship between these processes and patterns, and which are most important across contexts, is critical for effective management.

Beyond exploring processes that drive MPP patterns, the *plastic-scape* must also integrate the pattern-process relationships of MPP impacts on human and natural communities. Considering both patterns within the human and natural components of the *plastic-scape* can provide insight into the processes that drive patterns of impacts. For instance, overlaying patterns of MPP and human use of seascapes (e.g., tourist beaches or fishing areas), may inform patterns of high MPP impact (Beaumont et al. 2019; Leggett et al. 2014; Mouat et al. 2010). Currently, this is a significant gap in MPP research, which would benefit from place-based, seascape ecology approaches. Ultimately, as the impacts of MPP drive action, these pattern-process relationships should be centered in management approaches.

Scale and hierarchy organization. Scale is the grain (finest resolution) and extent (total area) of a seascape. As scale changes, dominant processes and patterns change (Wu 2012). To fully understand the *plastic-scape*, processes and patterns must be studied across spatial and temporal scales, and the correct scale for analysis will depend on the patterns or processes of interest (Figure 2).

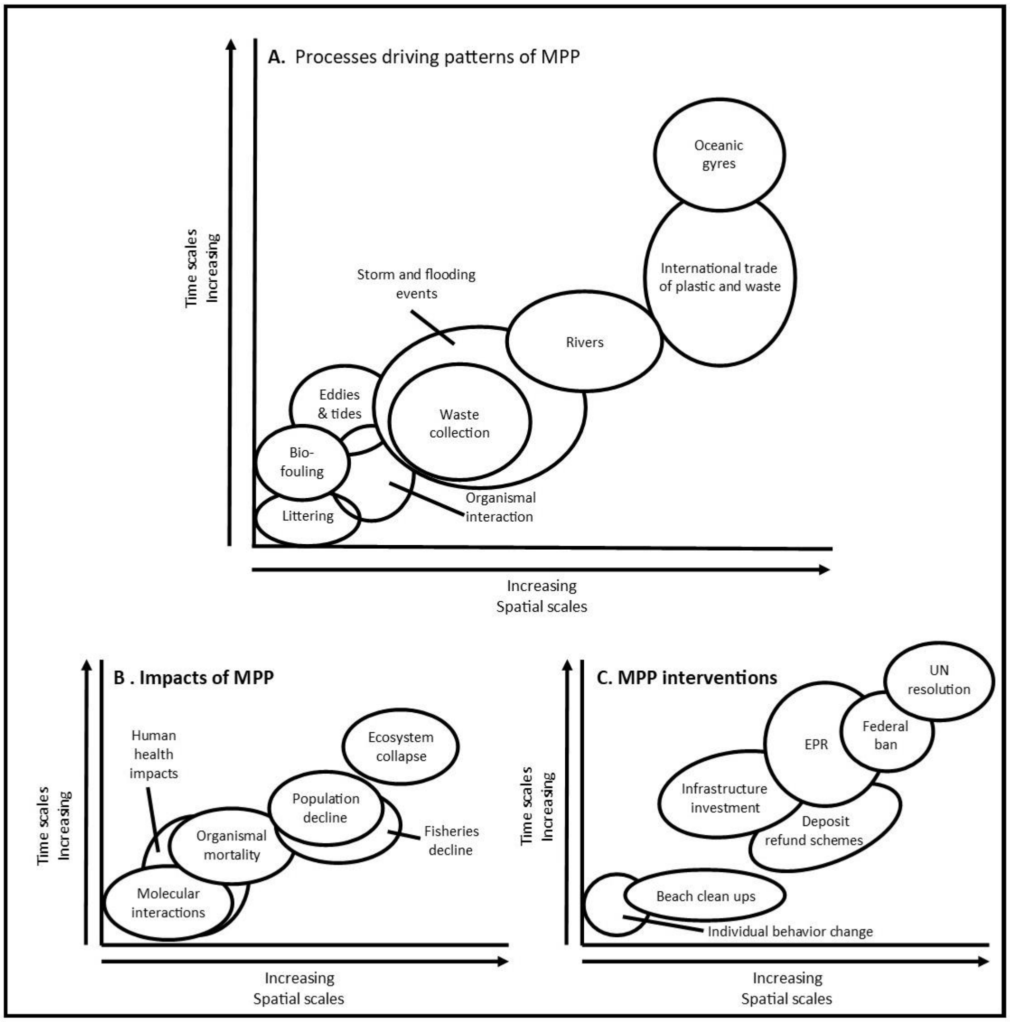

Figure 2. Time-space diagrams showcasing the multi-scale nature of the plastic-scape with spatial scale increase on the x axis (<mm to km) and time increasing on the y axis (<second to years). Circle length and width represent that relative range of spatial and temporal scale for a given dimension of the plastic-scape A: Provides examples of anthropogenic and natural processes that drive the spatial arrangement of marine plastic pollution. B: Examples of the social-ecological impacts of plastic pollution. C: Examples of marine plastic pollution management strategies.

At the global scale, particular nations have been identified as MPP sources, but at finer scales different leakage patterns emerge, such as high MPP densities near urban centers, rivers, and landfills (Eriksen et al. 2014; Huang et al. 2020). The dominant processes driving national leakage patterns are wealth, governance, and socio-economic status, while infrastructure, municipal management practices, and local hydrology are more important locally (Jambeck et al. 2015; Lebreton et al. 2017, Thushari and

Senevirathna 2020). Spatial and temporal scales are often linked, with change occurring faster at finer scales (Westley et al. 2002). Current-driven accumulation of MPP in oceanographic gyres is a global pattern-process relationship occurring on the time scale of years to decades, while finer scale patterns are driven by smaller and faster oceanographic processes—wave action, eddies, or tides (Brignac et al. 2019; Eriksen et al. 2014).

Hierarchy theory assumes systems can be divided into nested levels, where patterns and processes occurring across scales are part of a single system with cross-scale effects (Allen and Starr 2017; Kavanaugh et al. 2016).

Patterns and processes that emerge at different temporal and spatial scales of the plastic-scape influence each other. For example, global oceanographic processes are the dominant processes driving patterns of MPP associated with the gyres. However, these currents also contribute to local heterogeneity, such as the variation in MPP density between windward and leeward coasts (Brignac et al. 2019). Another cross-scale impact is the influence of national governance and socio-economic status on local plastic waste management strategies. National governance and wealth influence the resources, technology, and funding available to implement local waste management, ultimately changing local leakage rates (Helm et al. 2022).

Hierarchy theory can also be implemented to understand management across scales. Policies introduced at one scale of governance will influence others. For example, China's National Sword Policy, which regulates the import of recyclables, affected U.S. municipalities by decreasing the demand for plastic waste, ultimately driving local action (Murphy et al. 2020; Vedantam et al. 2022). Therefore, hierarchy theory provides an approach for understanding the *plastic-scape* as a whole and understanding the influence of interventions across levels of governance.

Methods and tools. SE provides tools, metrics, and methods that can be applied to the *plastic-scape* (Costa et al. 2018; Wedding et al. 2011). Additionally, it provides an ecological framework, technical skills, and best practices for applying them (Cumming et al. 2022; Grober-Dunsmore et al. 2009; Lepczyk et al. 2021).

Seascape ecologists employ a breadth of imaging tools—satellites and aerial photography, drones, boat-based sensors (e.g., LIDAR), autonomous vehicles, underwater imaging, benthic mapping, and semi-automated image classification—that can be used to map and monitor plastics (D'Urban Jackson et al. 2020; Costa et al. 2018). However, their limited use has focused on characterizing MPP transport and deposition (Lebreton et al. 2017; Salgado-Hernanz et al. 2021). MPP researchers have already called for the broader application of these methods, in the form of the integrated marine debris observing system, to develop global MPP maps for long-term monitoring and management (Maximenko et al. 2019).

SE also provides metrics to quantify characteristics of the *plastic-scape*. Spatial pattern metrics are applied to maps to quantify, characterize, and interpret patterns and pattern-process relationships (Boström et al. 2011; Pittman et al. 2021; Wedding et al. 2011). These metrics can be applied to the *plastic-scape* to quantify and interpret the distribution of MPP, the configuration of its social-ecological consequences, and the effects of management on these patterns.

Finally, SE provides modelling approaches. Network models, predictive spatial models, neutral seascape models and dynamic models have been applied to better understand marine conservation issues, characterize complex connectivity patterns at management appropriate scales, and simulate management outcomes under various scenarios (Costa et al. 2018; Engelhard et al. 2017; Pittman et al. 2007; Stamoulis et al. 2018; Treml and Kool 2018; Wedding et al. 2019). I have seen the value of modeling MPP to understand patterns of MPP leakage (Borrelle et al. 2020; Lebreton et al. 2017). The application of SE models will improve the evaluation of interventions, provide spatially explicit outputs, and allow for multi-scale models.

Transdisciplinary research for management. SE transdisciplinary approaches can inform more effective MPP research and management (Wu 2021; Pittman et al. 2021). First, research agendas should be co-produced. In SE, practitioners are being included in discussions about future research agendas, with their priorities deemed equally important to academics (Pittman et al. 2021). Though differences between these two groups remain, areas of agreement provide clear opportunities for collaboration (Cvitanovik et al. 2016; Dey et al. 2020). Setting a co-produced research agenda presents an opportunity for aligning the goals of the diverse group of stakeholders addressing MPP.

SE also provides methods for transdisciplinary research, including management specific metrics, predictive models to inform decision making, monitoring approaches, and tools to evaluate management outcomes (Naussauer and Opdam 2008; Olds et al. 2016; Pittman 2018; Pressey and Bottrill 2009). The benefits of these approaches are exemplified by their rapid adoption in biodiversity conservation, restoration, and sustainable development (Balbar and Metaxas 2019; Choi et al. 2008; Opdam et al. 2018).

Future Research

Generally, an SE approach should be applied to answer spatially explicit, place-based questions about patterns in the *plastic-scape,* and the processes that drive them, with a focus on informing management. Since MPP is primarily land-based, characterizing connectivity between terrestrial and marine systems is critical. Hydrological models have already been applied to identify MPP leakage patterns and particular rivers as management priorities (Correa-Araneda et al. 2022; Lebreton et al. 2017; Windsor et al. 2019). Future research could explore different scales and processes to identify other contributors to leakage patterns.

Researchers should also explore how seascape configuration influences MPP pathways and patterns. For example, certain habitats act as plastic sinks (Sanchez-Vidal et al. 2021; Martin et al. 2020). Research on the relationship between seascape configuration and MPP deposition can be used to predict MPP patterns and inform management priorities.

Future work could also employ social sensing—the characterization of human components of the *plastic-scape* (Liu et al. 2015). Integration of human activity and social data into MPP maps and models could provide more insight into anthropogenic pathways of MPP leakage and the efficacy of different management efforts.

Finally, research to inform and evaluate management should be prioritized. For example, researchers can employ predictive spatial models to compare outcomes associated with various intervention strategies and inform multi-scale research and management across different levels of governance. SE approaches could also provide baselines, allowing researchers to better monitor changes in *plastic-scape* patterns to evaluate management efficacy (Maximenko et al. 2019).

Limitations

Using the tools of SE, researchers can better understand the *plastic-scape*; however, this approach has limitations. The primary limitation is technological. To date, remote sensing has only been used to quantify surficial MPP (Goddijn-Murphy and Williamson 2019). Additionally, satellite data typically has a resolution of >1 meter, which is too coarse to detect most MPP. Though alternatives exist, they can be expensive (e.g., aerial imaging and high spectral sensors), inconsistent (e.g., thermal infrared sensing), or range limited (e.g., drones) (Goddijn-Murphy and Williamson 2019; Salgado-Hernanz et al. 2021). However, as technology improves and data collection becomes easier, the value of employing the SE approach will continue to increase.

Second, land-based pollution is not a research priority in SE (Pittman et al. 2021). Further, plastic pollution is a non-point source pollution with a complex life cycle largely driven by human activity (Napper and Thompson 2020). Identifying the appropriate scope and scale of analyses and actions may prove challenging. MPP also represents a breadth of pollutants that have different patterns, processes, and social-ecological consequences as they degrade, making MPP less predictable than other pollutants (Eriksen et al. 2014; Luo et al. 2022).

Finally, more research is needed on integrating human dimensions (e.g., ecosystem services) into SE models (Barbier and Lee 2014; Pittman et al. 2021). Still, LE and SE continuously adapt to better address applied research questions. Therefore, as SE is further applied to MPP research and management, many of these limitations could be addressed.

Conclusion

The *plastic-scape* includes all the human (i.e., governance systems and actors) and ecological components (i.e., abiotic, and biotic processes) that contribute to patterns of plastic production, use, and pollution, as well as the interactions between MPP and human and natural communities that drive its social and ecological consequences. Failures to effectively mitigate MPP and its consequences are exacerbated by the complexity of this system and the ad hoc, reductive nature of many research and management efforts. SE provides concepts, methods, and tools that can provide an approach for researching the *plastic-scape* and effectively mitigating MPP.

CHAPTER 3

A DECISION SUPPORT FRAMEWORK FOR ESTIMATING THE COST OF MARINE PLASTIC POLLUTION INTERVENTIONS

Introduction

Marine plastic pollution has detrimental effects on the environment, the economy, and human well-being (Beaumont et al., 2019). Recognizing the implications of this environmental problem, stakeholders—policy makers, nonprofit organizations, and businesses—have made significant investments to address plastic pollution. For instance, financial pledges at the 2017 Our Ocean Conference totaled $8.5 billion (Our Ocean, 2017) (this represents all pledges and all costs converted to 2019 U.S. dollars). However, funds for conservation efforts are limited, and these commitments have not sufficiently reduced marine plastic pollution and its ecological and social effects (Borrelle et al., 2020). To ensure these investments achieve the desired results and are economically viable, it is necessary to systematically evaluate the cost-effectiveness of interventions before implementation (Murdoch et al., 2007).

Identifying the cost-effectiveness of plastic pollution interventions requires understanding the cost of an intervention, its efficacy, and the benefits it produces (Cook et al., 2017). Current evaluations for effectiveness of plastic pollution interventions are insufficient (Löhr et al., 2017). Still, there are strategies for measuring the effectiveness of conservation policies (Sutherland et al., 2004) that could be applied to plastic pollution interventions. The literature on costs for conservation efforts, however, is sparse, and key costs are often omitted (Iacona et al., 2018), making it difficult to inform cost analyses for plastic pollution interventions. Most evaluations of plastic pollution interventions consider only the direct costs of intervention and recovered costs (e.g., taxes) to generate revenue (e.g., Crawford, 2008). Some consider the financial or nonmonetary benefits of plastic removal (Lavee, 2010), but many costs and benefits remain overlooked. Further, the inconsistent characterization and reporting of costs make it difficult to interpret studies or use them to inform decision-making (Iacona et al., 2018).

The challenge of standardizing the cost of interventions for plastic pollution is exacerbated by the breadth of intervention types. Interventions are implemented along the entirety of the plastic life cycle, yet cost analyses are only available for a small subset of these—such as cleanups (Mouat et al., 2010), deposit refund schemes (Lavee, 2010), and plastic bag bans (Zhu, 2011)—and analyses are predominantly conducted after implementation (Oosterhuis et al., 2014). Generalizable evaluations are complicated by the fact that the costs and possible benefits of interventions are influenced by factors specific to the context in which they are implemented (Oosterhuis et al., 2014). Different interventions place the burden of costs on different stakeholders. This is especially salient for marginalized populations, who are often disproportionately affected when the full distribution of costs is ignored (Adams et al., 2010). Thus, an approach for estimating the net costs of plastic pollution interventions is critical for helping decision makers better prioritize actions to achieve their conservation goals (Wilson et al., 2009).

I developed a decision support framework to identify the costs and benefits of plastic pollution interventions accrued by a range of stakeholders. I first identified the relevant categories of costs and benefits associated with plastic pollution interventions. I then used an equation to calculate the net cost as a function of these categories. I applied the framework to two quantitative case studies informed by specific interventions and four comparative case studies informed by the literature. Finally, to encourage more equitable decision making, I examined how context influences the distribution of costs across stakeholders. I sought to provide an approach to estimate and compare the costs of a range of interventions across sociopolitical and economic contexts.

Methods

A conventional cost–benefit analysis sums the benefits and subtracts the costs to yield the net benefits. However, my approach follows and extends on methods developed by Iacona et al. (2018), who examined the total costs of conservation interventions. I developed an equation to describe the net cost of mitigating marine plastic pollution, which I used to inform the development of my framework. Net cost is equal to the cost of implementing an intervention (direct, indirect, and nonmonetary [NM]) minus recovered costs and benefits (monetary and NM) produced by the interventions. If the sum is positive, there is a net cost. If it is negative, there is a net benefit:

$$\text{Net cost} = (\text{direct costs} + \text{indirect costs} + \text{NM costs}) - (\text{recovered costs} + \text{monetary benefits} + \text{NM benefits}) \qquad (1)$$

The cost and benefit categories were informed by a Web of Science search using a combination of the following terms: “cost” OR “economic” AND “marine” OR “ocean” AND “debris” OR “litter” OR “plastic.” I supplemented this with a Google Scholar search for gray literature (Appendix A).

Direct costs represent the costs of actions required to implement the intervention (National Center for Environmental Economics, 2010). There are four categories: overhead, labor, capital assets, and consumables (Iacona et al., 2018). Indirect costs are associated with the intervention but not directly tied to the financial cost of implementing actions, such as the opportunity cost of volunteers (National Center for Environmental Economics, 2010). Recovered costs are the revenue created by the intervention to reduce net costs. They are categorized as direct costs, which are implemented to reduce the implementer's cost, or indirect costs, which may benefit other stakeholders. Monetary benefits are the savings that would be accrued by stakeholders due to resulting reduction in marine plastic pollution. There are two categories of monetary benefits: benefits to marine sectors and healthcare savings (McIlgorm et al., 2011; Mouat et al., 2010; Newman et al., 2015). Nonmonetary costs represent the nonfinancial costs of an intervention (e.g., environmental tradeoffs), and the nonmonetary benefits represent the nonfinancial benefits of implementation. Nonmonetary costs and benefits are categorized as environmental or social (McIlgorm et al., 2011; Newman et al., 2015). Table 1 provides examples of each cost and benefit.

Table 1

Cost and Benefit Categories for Marine Plastic Pollution Interventions

Positive costs	Negative costs and benefits
Direct costs:	Recovered costs:
1. Overhead (e.g., administration, disposal)	1. Direct (e.g., taxes, fines, fees)
2. Labor (e.g., salaries, benefits, insurance)	2. Indirect (e.g., job creation, substitutes)
3. Capital assets (e.g., infrastructure, vehicles)	
4. Consumables (e.g., materials, gasoline)	Monetary benefits:
	1. Decreased cost of marine/coastal activities
Indirect costs:	a. Fisheries (e.g., propeller entanglement)
1. Opportunity cost (e.g., volunteer time)	b. Shipping/yachting (e.g., obstruction)
2. Job loss	c. Aquaculture (e.g., blocked pipes)
3. Substitution (e.g., alternative products)	d. Agriculture (e.g., coastal agriculture)

	e. Increased revenue in recreation f. Increased provisioning of marine resources 2. Reduced healthcare costs (e.g., injuries)
Non-monetary costs: 1. Environmental impacts of intervention 2. Social impacts of intervention	Non-monetary benefits: 1. Social benefits a. Human welfare (e.g., sense of place) b. Social justice (e.g., reduced inequity) 2. Environmental health (e.g., intrinsic value)

The framework provides a section for users to input the intervention’s description, its objectives (i.e., the primary goals of the intervention), and the spatial–temporal scale of evaluation (Table 2). Then, users record the stakeholders involved in or affected by the intervention. To identify the costs accrued by a specific stakeholder group, each stakeholder is listed in a new row. Next, the user evaluates each of the cost and benefit subcategories, as outlined below. Nonmonetary costs and benefits should be identified, even if users cannot estimate their monetary value because they often relate directly to the intervention objectives. The user can quantify them with nonmonetary units (e.g., number of animals saved). If the user wants to further enumerate nonmonetary costs and benefits, there are methods for doing so, such as ecosystem service accounting (Crossman et al., 2012). The final section provides an opportunity for users to conduct an equity evaluation, in which users identify stakeholders who would benefit or be harmed by each intervention and list net costs accrued by each stakeholder group.

Table 2

Decision Framework to Identify All Costs and Benefits Associated with Marine Plastic Pollution Intervention

Intervention: Description of the intervention
Objective: The overall goals of the implementing party
Scale: Spatial and temporal scale (e.g., municipality or nation; 1 year or 2 decades)

Stakeholders[a]	Actions and direct costs	Indirect costs	Recovered costs	Monetary benefits
Actors and those Affected (e.g., NGO, the public, government)	Steps to intervention and associated costs (e.g., enforcement, infrastructure)	Not associated with direct action (e.g., job loss, opportunity cost)	Direct or indirect revenue from implementation (e.g., fines, job creation)	Savings from plastic reduction (e.g., increases tourism)

[b]Non-monetary costs (e.g., environmental trade-offs, social costs)
Non-monetary benefits (e.g., ecosystem services, human welfare)
Equity: payers vs. beneficiaries

[a]In this section each row will hold one stakeholder group. The subsequent columns will list the actions, direct costs, and all other costs for each stakeholder group. [b]Non-monetary costs and benefits should be listed here with rows for each stakeholder group, but likely be unquantified.

Quantitative case studies. To demonstrate how the framework can be used to examine relative costs of alternative interventions, I applied it to two cases in which comprehensive cost evaluations were completed prior to intervention implementation. These cases allowed us to explore different interventions implemented by different actors under contrasting socioeconomic conditions.

The first case study explored implementation of a solid waste management (SWM) plan in the city of Bayawan, Negros Oriental, Philippines. The Philippines is ranked as the third largest producer of plastic pollution in the world, and plastic pollution has been found in the guts of marine species, including commercially important fish (Bucol et al., 2020). Bayawan is a 700-km2 coastal city on the island of Negros with a population of 117,900 (Philippines Statistics Authority, 2015). I explored the cost of implementing a 10-year SWM plan in Bayawan. My examination was informed by the public document, Solid Waste Management Plan (2019–2028). The key objectives of the plan were to expand waste management services, increase recycling and composting rates, and reduce open burning to ensure the city is prepared for anticipated population growth and urbanization. Key stakeholders for implementation include the municipal government, the local community, schools, barangays (neighborhoods), and industry (marine sectors and recycling sectors). The city identified actions required to achieve these objectives: purchase more equipment, build a new special waste facility, build a water monitoring pond, implement and enforce new SWM ordinances, support the establishment of barangay-based SWM facilities, and administer school education and innovation programs (City of Bayawan, 2019).

The second case study explored the implementation of a trash interceptor at the mouth of the Jones Falls River in Baltimore over a 10-year evaluation period (Clearwater Mills, 2013). Baltimore is a large coastal city—population of 593,490—in Maryland, USA (United States Census Bureau, 2019). It is located on the Chesapeake Bay, an ecologically and socially important body of water that is negatively affected by large amounts of plastic debris and microplastic pollution (Hale et al., 2020). The Waterfront Partnership is a group of businesses that agreed to pay additional taxes into a fund for cleaning up the waterfront. I obtained cost information from the Waterfront Partnership and the CEO of Clearwater Mills, the company that built and maintains the trash wheel. Cost data were provided at the project level and focused predominantly on the cost to the Waterfront Partnership. The key objectives of the trash wheel were to improve the sanitation and water quality of Baltimore’s Inner Harbor. Key stakeholders for implementation were the Waterfront Partnership, the city of Baltimore, the public, and a local marina. Actions taken to achieve the objectives were constructing, operating, and maintaining the trash wheel and educating the public (correspondence with Clearwater Mills and Waterfront Partnership).

Comparative case studies**.** To better understand three key factors that influence the net costs of intervention—temporal scale of analysis, spatial scale of implementation, and socioeconomic condition—I developed four conceptual case studies. In these case studies, I compared the costs of interventions in scenarios that varied one of these factors, while holding all others constant. I explored the influence of temporal scale on costs in case studies on a time scale of one year and 20 years. For the former case, I evaluated the costs of beach cleanups in developed municipalities, and for the latter I evaluated waste-to-energy (WTE) plants in developed municipalities. I explored the influence of spatial scale of implementation by comparing the costs of beach cleanups at the municipality and national scale in a developed country over one year. Finally, I explored the influence of socioeconomic conditions by comparing the costs of a WTE plant in a municipality in a developed versus developing country.

The choice of scenarios for each comparative case study was based on the availability of peer-reviewed and gray literature evaluating interventions with the appropriate socioeconomic conditions and spatial–temporal scale. I characterized all costs and benefits identified in the literature review based on the categories in my cost–benefit framework. I then identified how the relative costs for each of the cost and benefit categories differed based on the case study scenario (e.g., identified whether direct costs were higher or lower for beach cleanups or WTE on a 10-year time scale) (details available in Appendix A). To standardize comparisons across case studies, I assumed effectiveness was consistent for all interventions in a scenario (i.e., a bag ban implemented in a developing country and a developed country will reduce bag use by the same proportion).

Results

Implementation of a solid waste management plan in Bayawan**.** Based on available information, the net cost estimate for Bayawan over 5 years was $1,154,526 (Table 3). This was the direct costs of the program minus the costs recovered by fees, fines, and sale of recyclables. This estimate did not include indirect costs or monetary benefits, which would increase and decrease net cost, respectively.

The cost to the public was calculated as $38,600 ($0.33 per capita), which was the direct costs of fees and noncompliance fines. This estimate did not include the direct costs of purchasing waste-segregation containers, indirect costs, or nonmonetary costs, which would increase net cost. It also did not include recovered costs or benefits (monetary or NM), which would decrease net cost.

The benefit to schools was $52,110 based on the administration of government awards for the best waste management programs (net cost is negative). Importantly, these recovered costs would not be evenly distributed across schools but would benefit only schools deemed most innovative. This estimate also did not include the direct and indirect costs of implementing waste management plans in schools, which would increase costs. Also not included were additional recovered costs, such as the sale of recyclables, which would further reduce costs. Cost estimates were not available for barangays, the recycling sector, or marine sector.

The partial distributions of costs suggested the cost of this plan would fall primarily on the city. The benefits would be greatest for marine sectors, the recycling sector, and the public. The direct costs to the public appeared to disproportionately affect low-income, rural communities that historically burned or dumped waste at no cost and must either manage waste according to new ordinances or pay fines. Some low-income individuals could experience reduced income due to fewer opportunities for waste picking.

Table 3

City of Bayawan Case Study

Intervention: Implement mandatory waste segregation and collection throughout the city
Objective: Expand waste collection in all barangays and achieve 70% waste diversion
Scale: City of Bayawan, Negros Oriental, over 5 years

Stakeholder[a]	Actions and direct cost[b]		Indirect and nonmonetary cost	Recovered cost, monetary benefit, nonmonetary benefits	
City	Total capital assets:	$247,040	Indirect costs:	Recovered costs:	-$38,600
	Two garbage compacters	($231,600)	OC[d] of SWM committees	Tipping fees	
	Special waste facility	($9,650)		Dumping fines	
	Water monitoring pond	($5,790)		Garbage stickers	
				Recyclables sales	
	Total administration:	$946,086			
	Enact new SWM ordinances	($386)			
	Enforce SWM ordinances	($162,120)		Monetary benefits:	
	School innovation program	($52,110)		Clean-up[c]	
	Collection operations	($248,970)		Tourism[c]	
	Operation of BCWMEC facility	($451,620)			
	Expansion of SWM coverage	($30,880)			
Public	Purchase waste containers		Indirect costs:	Recovered costs:	
	Composting		OC[d] of waste segregation[c]	Recyclable sales	
	Payment of fees/fines	$38,600	Loss of informal waste sector[c]	Monetary benefits:	
			Nonmonetary costs:	Healthcare costs[c]	
			Environmental costs[c]	Nonmonetary benefits:	
				Human welfare[c]	
				Ecosystem health[c]	
Schools	Purchase of waste containers		Indirect costs:	Recovered costs:	
	Payment of fees/fines		Plastic alternatives[c]	Government awards	-$52,110
	Manage compost and MRF facilities			Recyclables sales	
Barangays	Collect/compost biodegradables				
	Enforce SWM ordinances				

Marine sector		Monetary benefits: Interaction costs[c]
Recycling sector		Recovered costs: Sale of recyclables

Net costs: Government: $1,154,526. Missing costs include indirect costs and monetary benefits. Public: $38,600 or $0.33/capita. Missing costs include some direct costs, indirect costs, recovered costs, monetary benefits, nonmonetary costs, and nonmonetary benefits. Schools: -$52,110. Missing costs include direct costs, indirect costs and more recovered costs. Barangays: Costs are not available. Missing costs include direct costs. Recycling sector: Costs not available. Missing costs include recovered costs. Marine Sector: Cost data not available. Missing costs include monetary benefits.

Equity: Costs are negative for industry and the public, and positive for the city, barangays, and schools. This may disproportionately affect low-income communities that could be burdened by waste-segregation costs and rural communities that receive fewer services from the city and have higher burdens for at-home composting and waste management.

[a] Includes the city, the public, schools, barangays, and industry (recycling and marine sectors).
[b] All costs are in 2019 U.S. dollars. Costs included without an estimate were mentioned in the report but not considered as costs. Costs in parentheses represent a subcost of the cost listed.
[c] Costs identified by the authors but excluded from the city's report.
[d] OC is an abbreviation for opportunity cost.

Implementation of a trash wheel in Baltimore. Net cost to The Waterfront Partnership over 10 years was $2,250,202 (Table 4). This was based on the direct cost of implementing the trash wheel minus costs recovered through financial support from the city, sale of trash wheel memorabilia, and tours of the trash wheel. This estimate did not include most recovered costs, monetary benefits, or nonmonetary benefits that would decrease net cost.

The cost to Baltimore was $619,900 and included the direct costs for operation and maintenance and the dumpster disposal fee. This did not include monetary and nonmonetary benefits that would decrease net costs. The primary monetary benefit to the city was reduced cleanup costs and the main nonmonetary benefits were positive perceptions and aesthetic values.

The cost to the marina was $21,600. This was the indirect cost of providing a slip for the vessel at half price. This estimate did not include the benefits gained by the marina. Finally, an estimate was not available for the cost to the public, but they accrued costs and benefits as well. The monetary benefits to the public were reduced healthcare costs. The nonmonetary costs were the environmental costs of waste collection, and the nonmonetary benefits were the improvements to human welfare and environmental health. Overall, every stakeholder group felt they benefitted from implementation of the intervention.

Table 4

Baltimore, Maryland Trash Wheel Case Study

Intervention: Establish a trash wheel at the mouth of the Jones Falls River
Objective: Clean up Baltimore harbor
Scale: City of Baltimore, Maryland, USA; 10 years

Stakeholder[a]	Actions and direct cost[b]		Indirect and nonmonetary cost	Recovered cost, monetary benefit, nonmonetary benefits	
Waterfront partnership	Overhead:	$54,000		Recovered costs:	
	Total capital assets	$704,000		Funds from Baltimore	($50,000)
	Floating platform	($113,400)		Sale of memorabilia	
	Waterwheel	($19,400)		Trash wheel tourism	
	Conveyer	($48,000)		Monetary benefits:	
	Power transmission	($22,700)		Increased tourism	
	Solar panels	($58,300)		Higher property values	
	Covering structure	($147,900)		Less public interaction	
	Controls/sensor	($13,000)		Nonmonetary benefits:	
	Pump system	($20,500)		Positive perceptions	
	Dumpster float	($52,900)			
	Debris rake system	($13,000)			
	Log lift system	($9,700)			
	Miscellaneous expenses	($7,600)			
	Installation	($77,700)			
	Service vessel modification	($19,400)			
	Facilities, equipment	($79,900)			
	Total labor	$1,217,100/10yrs			
	Insurance	($43,700/yr)			
	Monitoring	($19,400/yr)			
	Maintenance	($10,400/yr)			
	Dumpster transport	($37,400/yr)			
	Communications	($10,800/yr)			

	Total consumables	$325,102/10yrs)			
	Vessel operations	(65,702/10yrs)			
	Fuel	($3,200/yr)			
	Registration	($162+$54/yr)			
	Maintenance	($1,100/yr)			
	Slip fee	($2,200/yr)			
	Equipment expenses	($65,400/10yrs)			
	Fuel	($540/yr)			
	Maintenance	($1,100/yr)			
	Parts and materials	($4,900/yr)			
	Dumpster disposal	($194,000/10yrs)			
Public			Nonmonetary costs: Environmental[c]		Monetary benefits: Healthcare costs[c] Nonmonetary benefits: Human welfare[c]
Municipality	Operations & Maintenance	$500,000/10yrs			Ecosystem function[c]
	Funds to support WFP	($50,000/yr)			Monetary benefits:
	Disposal	$119,900/10yrs			Clean-up costs
	Disposal fees	($11,100/yr)			Nonmonetary benefits:
Marina			Indirect costs:		Positive perceptions[c]
			Slip donation	$21,600/10 yrs	Monetary benefits: Clean-up costs[c] Increased recreation[c]

Net costs: Waterfront partnership: 2,250,202. Missing costs include recovered costs, monetary benefits, and nonmonetary benefits. Public: Cost not available. Missing costs include monetary benefits, nonmonetary costs, and nonmonetary benefits. Municipality: $619,900. Missing costs include monetary benefits, and nonmonetary benefits. Marina: $21,600. Missing costs include monetary benefits.

Equity: Costs are negative for industry and the public, and positive for the city, barangays, and schools. This may disproportionately affect low-income communities that could be burdened by waste-segregation costs and rural communities that receive fewer services from the city and have higher burdens for at-home composting and waste management.

[a] Includes the city, the public, schools, barangays, and industry (recycling and marine sectors)

[b] All costs are in 2019 U.S. dollars. Costs included without an estimate were mentioned in the report but not considered as costs. Costs in parentheses represent a subcost of the cost listed.

[c] Costs identified by the authors but excluded from the stakeholder's reports.

Comparative case studies**.** The net cost of coastal cleanups in developed cities was larger when evaluated on a longer time scale (Morishige, 2010; Mouat et al., 2010; Stickel et al., 2012) (Table 5). Average annual direct costs were higher in the 10-year time scale because of anticipated increases in hourly wages and increases in plastic production and pollution that demand more hours of cleanup to achieve the same outcomes (Mouat et al., 2010; Stickel et al., 2012). Disposal costs also increased over time (Mouat et al., 2010). Generally, as landfill space decreased, disposal fees increased, and alternative disposal methods (e.g., controlled incineration) often had higher fees (Crawford, 2008). Monetary benefits decreased over the 10-year period because tourist expectations for cleanliness increase over time, which reduces the benefits of cleanups if effectiveness is held constant (Leggett et al., 2014; Mouat et al., 2010).

For WTE plants, net costs decreased as operational time increased (Crawford, 2008; Jamasb & Nepal, 2010). This was because of high direct costs. The most significant costs for WTE were capital assets, which are cheaper per annum the longer a plant operates (Lombardi et al., 2015). Some direct costs increased over time, such as operation, maintenance, and labor costs—due to increases in salaries (Crawford, 2008; Jamasb & Nepal, 2010), but capital assets dominated these other direct costs for WTE. The indirect costs of WTE also decreased with time. As technology and emission standards improved, the amount of air pollution released decreased, reducing human health costs. Decreased pollution reduced nonmonetary costs of WTE as well (Jamasb & Nepal, 2010). Energy capture also improved with advances in technology and quality of feedstock, which increased recovered costs through energy sales and increased nonmonetary benefits associated with reducing net greenhouse gas emissions (Crawford, 2008; Jamasb & Nepal, 2010).

Net costs of coastal cleanups were higher per unit cleaned when cleanups were implemented at the national level than at the municipal level (Morishige, 2010; Mouat et al., 2010; Stickel et al., 2012). Coastal cleanups implemented at the local level were most often carried out in popular tourist sites with sandy beaches (true for more than 90% of municipalities in the United Kingdom [Mouat et al., 2010]). Cleanups on these beaches had lower direct costs, including labor, transportation, and possible healthcare costs, because sandy beaches have lower plastic retention rates, are easier and safer to access, and are faster to traverse than rocky shores (Mouat et al., 2010). These beaches also provided higher monetary benefits because they received more recreational use (Morishige, 2010; Leggett et al., 2014). National-level cleanups would include a higher proportion of isolated coastlines and other shore types, such as rocky and muddy shores. Higher direct costs, including higher transport and labor costs for these regions, would raise the average cost per kilometer of coastline, whereas the monetary benefits to tourism and human health per kilometer cleaned would decrease.

The net cost of implementing a WTE plant was higher in municipalities in developing countries than in developed countries (Lombardi et al., 2015; Yang et al., 2012). Although labor costs were lower in developing countries (Kaza et al., 2018), infrastructure costs were higher for developing countries as a function of gross domestic production, making capital costs more prohibitive (Fobil et al., 2005). Additionally, WTE plants in developing countries typically used older technology and had waste with a higher moisture content, which affected several costs and benefits. This increased maintenance costs because waste with high moisture content generates more corrosive by-products that damage boiler tubes (Zhang et al., 2015). Indirect and nonmonetary costs were also higher because both older technology and high-moisture-content waste produced more air pollution and greenhouse gasses (Lombardi et al., 2015; Yang et al., 2012). Increased rates of groundwater contamination further elevated these costs because toxic ash must be put in a landfill (Kaza et al., 2018) and landfill leakage rates were generally higher in developing countries (Zhang et al., 2015). Finally, plants in developing countries produced less energy, which decreased recovered costs (Lombardi et al., 2015).

Table 5

Comparative Case Studies of Costs and Benefits of Plastic Pollution Interventions

Factor	Cost category	Comparative case studies[a]			References
Time Scale		Coastal cleanup, developed city			
		1 year		20 years	Balance et al. 2000, Han et al. 2010, Mouat et al. 2010; Sticketl et al. 2012, Leggett et al. 2014
	Direct	Labor	<	Labor	
	Direct	Diposal	<	Disposal	
	Avoided	Tourism	>	Tourism	
		Waste-to-energy, developed city			
		1 year		20 years	Crawford, 2008; Yang et al. 2012; Lombardi et al. 2015
	Direct	Maintenance	<	Maintenance	
	Indirect	Human health	>	Human health	
	Recovered	Energy sales	>	Energy sales	
	Nonmonetary cost	Pollution	>	Pollution	
	Nonmonetary benefit	Greenhouse gas sink	>	Greenhouse gas sink	
Spatial Scale		Coastal cleanup, developed locale, 1 year			
		City		Country	Balance et al. 2000, Han et al. 2010, Mouat et al. 2010; Sticketl et al. 2012, Leggett et al. 2014
	Direct	Labor	<	Labor	
	Direct	Transportation	<	Transportation	
	Direct	Disposal	<	Disposal	
	Monetary benefit	Human health	<	Human health	
	Monetary benefit	Tourism	>	Tourism	
Socio economic context		Waste-to-energy, 20 years			
		City, developed country		City, developing country	Dijkgraaf & Vollebergh, 2004; Consonni et al. 2005; Crawford, 2008; Fobil et al. 2005; Jamasb & Nepal, 2010; Lombardi et al. 2015; Yang et al. 2012; Zhang et al. 2015; Mavrotas et al. 2015; Xin-gang et al. 2016; Wang et al. 2016; Kaza et al. 2018
	Direct	Infrastructure	<	Infrastructure	
	Direct	Labor	>	Labor	
	Direct	Maintenance	<	Maintenance	
	Indirect	Human health	<	Human health	
	Indirect	Job loss informal sector	<	Job loss informal sector	
	Recovered	Energy sales	>	Energy sales	
	Nonmonetary cost	Environmental trade-offs	<	Environmental trade-offs	
	Nonmonetary benefit	Greenhouse gas sink	>	Greenhouse gas sink	

[a]Comparative case studies hold all constant except the factor shown in column 1. Differences between cost categories are identified as being relatively higher or lower than the case study of comparison. Cost categories are shown in column 2. Relative cost differences are informed by references provided.

Discussion

Many decision makers try to maximize efficiency through wise investment when they are implementing conservation interventions (Murdoch et al., 2007). However, most assessments fail to capture the full suite of costs and benefits associated with a given intervention. As a result, investments in conservation often fail to achieve their stated objectives. My framework provides an approach for evaluating the net cost of alternative interventions for mitigating marine plastic pollution and supports a more standardized and equitable assessment of costs and benefits. Employing my approach facilitates deliberation about the possible costs that may influence the efficiency of an intervention, allowing decision makers to compare an intervention to a business-as-usual scenario or other possible interventions before their implementation.

Decision makers can also use this framework to compare costs across locations. When costs are not fully considered or clearly presented in studies, it is difficult for decision makers to interpret these costs and understand how they may differ in their own context. My costing framework promotes consistency in costing and reporting that will also allow researchers to better study relationships between cost and efficacy and understand how implementation context affects cost.

Use of this framework can also help increase the equity of interventions by ensuring decision makers consider the full distribution of costs to stakeholders across time. Plastic pollution disproportionately affects marginalized and low-income communities (Newman et al., 2015). Unfortunately, many conservation interventions have high social costs as well (Adams et al., 2010). For instance, WTE plants are promoted as a solution to high levels of plastic pollution interaction for marine organisms (McKinsey & Company & Ocean Conservancy, 2015). However, their historic construction in marginalized communities' places higher health costs and nonmonetary costs on these individuals (UNEA, 2019). This framework enables decision makers to understand cost distributions across stakeholders, allowing them to choose more equitable interventions or implement secondary policies (e.g., benefit transfers) to reduce an intervention's burden on vulnerable populations. To ensure this objective is achieved, it is critical that decision makers use a participatory approach, engaging with a diverse group of stakeholders in the process of identifying and analyzing costs.

Key factors for cost. I identified three factors decision makers should consider with the implementation of interventions for plastic pollution: temporal scale of analysis, spatial scale (i.e., international, national, municipal) of implementation, and socioeconomic conditions. The net cost of a coastal cleanup per kilometer of beach cleaned at the municipality scale increased with time scale of analysis, whereas the net cost of a WTE plant decreased. This indicates the importance of the temporal scale of cost–benefit analyses when evaluating the feasibility of individual interventions and when comparing interventions. Some interventions, such as coastal cleanups, may be cost-effective when evaluated annually because of tourism benefits (Balance et al., 2000; Stickel et al., 2012). However, other interventions may achieve the same objective while being more cost-effective when evaluated on a longer time scale (de Araújo & Costa, 2006). Alternatively, WTE may be infeasible if considered on a short time scale, but many cities in developed countries have achieved net negative costs over the course of a few decades (Crawford, 2008). Notably, costs may shift again over time as waste streams change. There are developed countries that must now import feedstock waste to maintain their plants (Olofsson et al., 2005). Therefore, the temporal scale of analysis should be in line with the objective. If the objective is long-term sustainability, then the temporal scale of evaluation should be longer. Ultimately, it may be best for communities to implement multiple interventions that aim to achieve objectives with different time scales.

Spatial scale of implementation may significantly change the cost of an intervention; however, many interventions are advocated for across dramatically different scales of implementation. For example, plastic bag reduction policies are often implemented at the national level, but in the United States, where no federal policy has been implemented, hundreds of states and cities have implemented their own legislation (Giacovelli, 2018). Economies of scale can significantly influence the feasibility of conservation efforts (Armsworth et al., 2011). Before adopting policies that have been implemented at different scales, implementers should evaluate the cost of the intervention at their scale of implementation to ensure cost-effectiveness is not hindered.

Decision makers must also consider socioeconomic conditions when implementing interventions. Following the lead of the developed world, developing nations are investing heavily in WTE plants (UNEA, 2019). However, without external investment, low-quality technology may be implemented, which has detrimental impacts for ecosystem and human wellbeing (Lombardi et al., 2015; Yang et al., 2012). Additionally, indirect economic costs for local communities may be more severe in developing nations because WTE reduces the availability of high-quality waste for informal waste pickers (Kaza et al., 2018). Without consideration of the socioeconomic context, these interventions, which may be effective in certain countries, may be infeasible or detrimental in other contexts.

Recommendations for framework use**.** This framework should be used by any actor (e.g., municipality) considering the implementation of an intervention for marine plastic pollution. First, they should identify the objective of the intervention and the socioeconomic and environmental context of implementation. This information will help inform which interventions may be most effective, the time frame of consideration, and relevant stakeholders. Next, all key stakeholders must be identified and engaged early. Decision makers may be unaware of potential costs and benefits important to other stakeholders. A participatory approach will help ensure a complete assessment of costs and benefits. Finally, net costs can be quantified for each stakeholder group. Transparency throughout this process can help ensure costs are more equally shared and that social, economic, and environmental objectives will be achieved.

Hard to quantify costs and benefits. Many costs and benefits can be difficult to quantify— particularly indirect costs, nonmonetary costs, monetary benefits, and nonmonetary benefits. Decision makers can improve their estimates by applying other methods for quantifying costs and benefits in concert with my framework. For example, cost effectiveness analyses—first used in public health—can be used (Bojke et al., 2018). Additionally, methods such as ecosystem service valuation can be used to estimate the value of nonmonetary costs and benefits of plastic pollution interventions (e.g., Beaumont et al., 2019), but the lack of standardization in these approaches may create challenges for comparing values across studies and contexts (Seppelt et al., 2012).

Addressing data gaps. It will not always be feasible to quantify every cost and benefit for an intervention. In instances where costs and benefits cannot be financially quantified, other metrics can be used (e.g., animal deaths avoided) to inform decision-making. Additionally, decision makers can rarely identify all costs and benefits to each stakeholder group but must make the decisions with the data they have (Iacona et al., 2018). Therefore, systematic identification of costs and benefits to all stakeholders can improve the decision-making process

Considering long time horizons. Though I noted the importance of evaluating interventions on the appropriate time horizon, applying the framework over long time horizons requires additional consideration. First, quantifying costs is more difficult over long-time frames. Therefore, when considering an intervention, decision makers must acknowledge the uncertainty in expected cost estimates and anticipate realized costs may be greater. Additionally, costs and benefits accrue on different time horizons (O’Mahony, 2021). Therefore, when using the framework on a long-time horizon it is important to appropriately discount expected costs and benefits that are realized at different points in the future. This will allow the decision maker to make fairer comparisons across interventions in terms of their net present value.

Conclusion

In this chapter, I present a framework for evaluating and reporting the net cost of an intervention for marine plastic pollution. I developed this framework to help decision makers and researchers estimate the net costs of different intervention strategies before they are implemented, interpret cost estimates provided in other studies, and compare intervention costs and benefits across contexts. In using this framework, decision makers can identify the distribution of costs across stakeholders so that they can ensure interventions are equitable.

Ultimately, this will help ensure conservation efforts can be met with limited funds available for achieving these objectives. As research on the cost of plastic pollution and the efficacy of policy measures improves, it will strengthen the quality of the cost-benefit estimates the framework provides. Future research should seek to engage decision makers in various geopolitical and socio-economic contexts and at different scales of action to validate the efficacy of this tool and generate cost data that can be compared across contexts.

CHAPTER 4

A MULTI-TAXANOMIC, TRAIT-BASED FRAMEWORK FOR ASSESSING MACROPLASTIC VULNERABILITY

Introduction

Plastic pollution interactions have been recorded in hundreds of marine species (Bucci et al. 2020). As such, general approaches for assessing risk from both macro- and microplastics are urgently needed. Risk frameworks for microplastics are concentration-based and driven by effect via ingestion (Mehinto et al. 2022). For macroplastics (defined here as plastics > 5mm), frameworks identifying macroplastic concentration alone are less appropriate, as vulnerability is dependent on interactions beyond ingestion, such as entanglement and shading. Here, impacts likely vary based upon the characteristics of the plastic debris and the organism.

Interactions with macroplastic occurs primarily through ingestion, entanglement, or shading, and has been linked to injury, illness, and mortality (Bucci et al. 2020). However, 90% of studies evaluating impacts have measured effects at or below the organismal level (Bucci et al. 2020). Limited research has addressed the consequences of individual interactions with macroplastics on populations, communities, or ecosystems (Koelmans et al. 2017). For example, few recordings of macroplastic ingestion have been linked to population decline or adverse ecological outcomes (Bucci et al. 2020). Understanding these effects is critical for informing and prioritizing future research, management, and policy (Koelmans et al. 2017).

Trait-based approaches (TBAs) offer a method for inference across and within biological levels of organization and different geographies by comparing biological, ecological, and physiological characteristics that make an organism vulnerable to different stressors to estimate the relative risk of impacts on populations and communities (Van den Brink et al. 2011). Over the last few decades, TBAs have been applied to a breadth of stressors (e.g., pesticides, metals, pharmaceuticals, and petrochemicals), and are increasingly use to inform regulatory frameworks for ecological risk assessments (Van den Brink et al. 2011, Polidoro et al. 2021, De Lange et al. 2009, Golden and Rattner 2003). Early research on the applicability of TBAs for plastic pollution show promise. Good et al. (2020) applied a TBA to evaluate the vulnerability of marine birds in the California Current Large Marine Ecosystem and found that pelagic species are at greater risk than coastal species. Similarly, Compa et al. (2019) analyzed data from 26 studies representing 84 species from six taxa to identify traits associated with exposure to plastic ingestion. Both studies are limited, however, in the traits they consider. To standardize the use of TBAs for plastic pollution, a more comprehensive traits framework is needed to better estimate vulnerability for cross-taxa and cross-locale comparisons.

I present a trait-based framework that can be applied to estimate the relative vulnerability of marine species to the physical impacts of macroplastic pollution (>5mm). Due to its comprehensive nature, this multi-taxonomic framework can be applied to develop vulnerability indices of species within or across taxonomic groups from local to global scales. Application of this framework can be used to identify vulnerable marine species and communities for targeted management efforts, long-term monitoring, and more in-depth risk assessments.

Methods

To develop this framework, I first identified all traits that have been associated with increased species vulnerability to the physical impacts of macroplastic pollution. I focused on the physical impacts of macroplastic across all taxa because I found the impacts of microplastics, nanoplastics, and associated chemicals differ from macroplastics and should be considered independently of macroplastics (Koelmans et al. 2017). I then categorized traits from our literature review into three dimensions of population risk assessment to inform vulnerability: likelihood of exposure, species sensitivity, and population resilience (Polidoro et al. 2021).

Literature review. I identified traits through a comprehensive review of the literature from 1898 to 2021. I began my review with the literature presented in Bucci et al. (2020) (through Nov. 2017), only reviewing studies that included plastics greater than 5 mm in size. I then applied the same methods as those presented in Bucci et al. (2020) to search Scopus for literature from November 27th, 2017, to March 31st, 2021, using the terms "marine debris", "plastic debris", "macrodebris", and "mesodebris."

Each abstract was reviewed once to determine if the paper should be included in the review. Papers were excluded if they did not evaluate the physical effects of plastic pollution, exclusively evaluated microplastics (plastics <5mm), or did not present novel data (e.g., literature reviews, perspective pieces). For each paper included in the final review, I recorded the author, year of publication, taxonomic group and species evaluated, study location, exposure type (i.e., ingestion, entanglement, other), age of study individuals, information about the effect demonstrated, and any evidence of a relationship between a biological, physiological, or ecological trait and effect measured. I also collected information about the plastic material in the study (e.g., size, shape, polymer), the study itself (e.g., observational, experimental, marine,) and included a summary of the study.

Each effect measured was categorized by an effect type. If a study only evaluated frequency of plastic interaction occurrence, then the effect measured was designated as exposure. Other effect categories included, but were not limited to, body condition (e.g., emaciation), injury (laceration, gut perforation), mortality, population decline, or assemblage shift. A new effect line was coded for each effect-species combination in a study (i.e., if a study evaluated injury and mortality rates for two species, then four distinct effects were coded in our review—injury data for species one, injury data for species two, mortality data for species one, mortality data for species two). However, if studies evaluated assemblages (typically invertebrates), presented results for several species together, or evaluated one effect type for more than 20 species, I coded this as one effect line with the species coded as “multiple”. If effects of plastic interaction were evaluated for a species-study combination, I did not code frequency of occurrence independently. I simply provided this information in the study summary. (See Supplementary Materials for literature review results).

Categorizing traits. Traits identified to be associated with vulnerability were aggregated into broad trait buckets. For instance, a study finding dipping and seizing increased ingestion rates and a study finding diving decreased plastic ingestions rates, would both exemplified “feeding and foraging behaviors” as a trait influencing vulnerability. Once all the traits identified in the literature review were characterized, I then categorized them into three dimensions of vulnerability—likelihood of exposure, species’ sensitivity, and population resilience—informed by Polidoro et al. (2020). Categorization was informed by how the trait influenced vulnerability. Traits that increased the likelihood of a species having plastics in their proximity, were categorized in “likelihood of exposure”, traits that increased the likelihood of a species to interact with plastic in their surroundings and/or have negative outcomes from interactions were categorized as “species’ sensitivity” and traits that influenced population recovery to interactions with plastic pollution were categorized as “population resilience.”

Results and discussion

Literature review results. I reviewed a total of 212 unique studies. Seventy-seven considered entanglement, 137 considered ingestion and 24 looked at other interactions, including shading, use in nests, rafting, suffocation, and proximity (note several studies considered multiple interaction types). Of the 77 entanglement studies, 38 included mammals, 15 reptiles, 14 fish, 14 birds, 11 invertebrates and only one plants. These papers measured a variety of effects, but the three most evaluated were exposure (23.4%), injury (55.8%) and mortality (41.6%). Most studies focused on sub-organismal (64.9%), and organismal (58.4%) effects, with only 7.8% evaluating population level and 2.6% evaluating community level effects. Almost all studies (97.5%) reported an “effect measured” (i.e., found plastic interaction and/or negative impacts), while only 18% (14 studies) reported a total of 15 “no effects” (i.e., an effect was measured and not demonstrated). Note this does not equal 100%, because some studies measured multiple effects with some showing “effects measured” and others showing no effect (i.e., an effect can be measured for injury, but no effect measured for morality).

Of the 137 studies that evaluated ingestion, 52 included birds, 35 mammals, 28 mammals, 28 fish, and two invertebrates (Table 6). Like the papers evaluating entanglement, exposure (58.4%), injury (15.3%) and mortality (33.6%) were commonly evaluated; however, change in body condition was also frequently observed in ingestion studies (16.1%). Most studies focused on sub-organismal (29.2%), and organismal (83.9%) effects, with only 1.5% evaluating population level and 0% evaluating community level effects. Almost 90% of studies reported an "effect measured" (89.8%), but ingestion studies were more likely to report no effect—39 studies (28.5%) had a total of 62 "no effect measured".

Table 6

Evidence Found for Effects Measured by Interaction Type

Interaction type	Effect measured	Total	Yes[a]	No
Entanglement	Exposure	18	18	5
	Body condition	2	2	1
	Injury	43	43	1
	Speed	1	1	0
	Reaching the sea	1	1	0
	Nesting deterrent	1	1	0
	Nest distribution	1	0	1
	Mobility	1	1	0
	Disease	1	1	0
	Community shift	1	1	0
	Crawl obstruction	1	1	0
	crawl time	1	1	0
	Mortality	32	30	2
	Population decline	6	2	4
Ingestion	Exposure	80	79	14
	Body condition	22	19	4
	Food consumption	1	0	1
	Risk	1	1	0
	Injury	21	12	11
	Mortality	46	44	3
	Population decline	2	1	1

[a] Yes indicates there was evidence of the measured effect. No indicates there was not.

Traits associated with likelihood of plastic exposure**.** I identified seven traits that influence a species' likelihood of exposure to macroplastics, which must be considered along with environmental macroplastic concentration: distribution, water column position, habitat, longevity, motility, longevity of the most sensitive pre-adult stage, and distribution of the most sensitive pre-adult stage (Table 7). Distribution, water column position, and habitat influence the likelihood that a species encounters macroplastic in their environment, as species present in areas with higher densities of macroplastic will have a higher likelihood of encounter. For instance, plastic accumulates near coasts and in gyres (Eriksen et al. 2014). If a species' range overlaps with accumulation zones it may have a higher likelihood of plastic ingestion and entanglement. Proximity to coasts, urban populations, and anthropogenic activities have all been associated with increased exposure to macroplastic (Thiel et al. 2018). Similarly, patterns of different plastic densities throughout the water column, such as increased density on the ocean surface and/or seafloor, can inform likelihood of exposure (Choy et al. 2019), as benthic species or species that live at the surface may encounter more plastics than species in the middle of the water column (Mouchi et al. 2019, Raum-Suryan et al. 2009). Habitat preference provides a higher resolution of exposure as some habitats are depositional zones for accumulating more plastics. For example, macroplastics are more likely to accumulate in rocky substrates or marine canyons than on reef slopes (Corcoran 2015, Page et al. 2004).

Table 7

Likelihood of Exposure Component of the Macroplastic Vulnerability Index Framework

Likelihood of exposure							
Trait	Distribution	Water Column Position	Motility	Longevity	Habitat	Longevity of most sensitive pre-adult stage	Distribution of most sensitive pre-adult stage

Assumption	Species with more of their range overlapping with macroplastic accumulation areas have greater exposure	Species that spend more time where plastic accumulates in the water column will have greater exposure	Exposure rates differ between sessile, small-range, and large-range species	Longer-lived adults will have more repeated exposures	Certain habitats accumulate more plastics than others	Likelihood of exposure will increase with the longevity of the most sensitive pre-adult stage	Likelihood of exposure is increased due to pre-adult stage distribution or mobility
Example indicators	•Overlap with plastic accumulation zones •Proximity to human activity	•Zone (e.g., benthic) •Depth range	•Site fidelity •Mobility	•Lifespan	• Foraging habitat • Nest habitat	•Time in most sensitive pre-adult stage	•Overlap of pre-adults and plastic accumulation zones

Motility and longevity can also influence a species' likelihood of exposure to plastic pollution. Some studies found species with larger foraging ranges may be at higher risk of plastic exposure (Raum-Suryan et al. 2009). Alternatively, sessile, or nearly sessile species cannot escape plastic interactions, so they may be at higher risk of exposure in high accumulation areas. For instance, corals and sponges may be particularly vulnerable to entanglement (or smothering), since they are benthic organisms often found in coastal areas near urban zones (Mouchi et al. 2019). When applying the framework, the influence of motility on sensitivity may be bimodal or taxa dependent. Adults that are longer lived have more opportunity for repeated contact with marine plastic pollution over time.

For several species, distribution and longevity of the most sensitive pre-adult stage should also be considered, as adult and sub-adult life stages of many marine organisms occupy different ranges, habitats, and positions in the water column (Raum-Suryan et al. 2009). For instance, juveniles in many fish families inhabit nursery areas in estuaries or coastal waters that are kilometers to hundreds of kilometers from adult habitats (Gillanders et al. 2003).

Traits associated with species sensitivity to plastic. Species sensitivity refers to traits that influence variation in individual rates of interaction with plastic and physiological responses to plastic ingestion, entanglement, or shading, such as injury, reduced body condition, and mortality. In this study, I identified nine traits influence species' sensitivity to plastic—body morphology, feeding and foraging behavior, prey preferences, non-foraging behaviors, egestion potential, respiration mode, behavior of pre-adult stages, relative physiological susceptibility of pre-adult stages, and reduced fitness due to other stressors (Table 8). Importantly, species sensitivity to ingestion, shading and entanglement is also dependent on the type of macroplastics.

Table 8

Species Sensitivity Component of the Macroplastic Vulnerability Index Framework

Species sensitivity									
Trait	Body morphology	Feeding and foraging behaviors	Prey preferences	Non-foraging behaviors	Egestion potential	Respiration mode	Behavior of pre-adult stages	Relative physiological sensitivity of pre-adult stages	Reduced fitness from other stressors
Assumption	Certain morphologies will be more sensitive to macroplastic	Certain feeding and foraging behaviors increase macroplastic sensitivity	Certain prey preferences increase macroplastic sensitivity	Some non-foraging behaviors may increase macroplastic sensitivity	Species that can egest plastics are less sensitive to ingestion	Certain modes will be more sensitive to entanglement	Differences in pre-adult and adult behavior may increase macroplastic interaction	The most sensitive life stage will have the greatest influence on sensitivity	Species impacted by other stressors will be more sensitive to macroplastic
Example indicators	•Stomach: mouth ratio •Gape size •Body shape •Body size	•Active vs. passive •Feeding strategy (e.g. diving, dabbling)	•Prey type (e.g., fish, cephalopod) •Prey specificity	•Curiosity •Aggression •Nesting	•Ability to regurgitate •Ability to pass debris	•Presence or absence of lungs or gills	•Altricial vs. precocial young •Foraging behaviors •Curiosity	•Relative sensitivity of pre-adult stage to adult	•Proportion of range with high temps, urbanization, or hypoxia

Feeding and foraging behaviors may also influence species sensitivity to plastic. For example, feeding behaviors can influence rates of both ingestion and entanglement (Page et al. 2004, Bond et al. 2013). Surface seizing and dipping birds are at higher risk of ingestion, while divers are at lower risk (Roman et al. 2019, Bond et al. 2013). Alternatively, diving species are more likely to drown from entanglement in marine debris than surface seizers (Thiel et al. 2018). Scavengers and opportunistic feeders experience more plastic ingestion and entanglement, due to increased interaction with vessels, ports, dumps, and fishing gear—common sources of macroplastic (Thiel et al. 2018, Basto et al. 2019). Finally, the strategies organisms use to sense their prey may influence their likelihood of ingestion, such as sight, sonar, or smell (López-López et al. 2018). For instance, the smell of biofouled plastics attracted turtles through a similar mechanism as their food (Pfaller et al. 2020). Overall, the link between feeding and foraging behaviors and ingestion sensitivity were well-documented for marine vertebrates, with sea birds being the most well researched, but evidence for feeding and foraging behavior influencing entanglement sensitivity were also observed in marine vertebrates and invertebrates.

Prey preferences also influence species sensitivity. Generalists may be more likely to consume plastics than specialists (Francis et al. 2020). Predators are also more likely to consume plastics if common plastics resemble their prey; for instance, soft, white plastics resemble jellyfish and squid (Poli et al. 2015). Additionally, carnivores can be exposed through secondary ingestion, so if their prey eats more macroplastic, they could consume more (Romeo et al. 2015), while herbivores can consume plastics entangling plants (Guterres-Pazin 2012). Prey preferences can also increase the likelihood of detrimental outcomes, as certain plastics are more likely to cause impaction and perforation in the GI tract (Roman et al. 2019). Prey preferences can increase entanglement sensitivity, as certain prey types are more likely to be near entangling items, such as fishing gear. Fish-eaters, detritivores, or scavengers are more likely to seek out active and ghost fishing nets, which are common entanglers for marine vertebrates and invertebrates (e.g., crab) (Good et al. 2010). This can lead to disproportionally high mortality rates because fishing nets are more likely to cause death from entanglement than consumer plastics (Costa et al. 2020). In summary, prey preference was closely linked to several components of macroplastic sensitivity for many taxa, increasing likelihood of ingestion and effects from ingestion for marine mammals, birds, turtles, and fish; and increasing likelihood of entanglement for all marine vertebrate orders, and many marine invertebrates.

Non-feeding behaviors also influence species sensitivity to plastic. Curiosity and aggression have both been linked with higher rates of plastic ingestion and entanglement in marine mammals (Raum-Suryan et al. 2009). These behaviors can be sex-linked in certain species, with research indicating that males may be more vulnerable in pinnipeds (Dau et al. 2009). Nesting behaviors have been shown to influence species sensitivity as well (Townsend and Barker 2014). Some bird species preferentially select plastics for nest building, increasing their own sensitivity to entanglement as well as their offspring's sensitivity (Townsend and Barker 2014). The link between non-foraging behaviors and plastic ingestion and entanglement were only documented in marine mammals and birds, with a focus on specific behaviors, but it is possible non-foraging behaviors influence sensitivity for other species as well.

Egestion potential influences a species' sensitivity to the physical impacts of plastic ingestion. Lower plastic accumulation rates have been observed in species that can regurgitate or easily pass consumed plastics, such as gulls (Basto et al. 2019), compared with species that cannot easily egest plastic once it is consumed, such as storm petrels and sea turtles (Wilcox et al. 2018, Nam et al. 2021).

Mode of respiration also influences sensitivity to entanglement. Air breathing species are more vulnerable to entanglement than non-air breathing species due to risk of drowning (Thiel et al. 2018, Dau et al. 2009). Fishes may be injured or hindered but are less likely to die quickly from entanglement (Nunes et al. 2018).

Both the behavior and relative physiological susceptibility of pre-adult stages are also important, as the behavior and morphology of pre-adults vary from adults for many species. For many species, the inexperience of young animals has been associated with higher ingestion and entanglement rates than for adults (Page et al. 2004, Costa et al. 2020). Juveniles may be more likely to mistake plastics for food items (Ryan et al. 2016). Additionally, young pinnipeds are often more playful than adults and as a result may have higher entanglement rates (Raum-Suryan et al. 2009). The physiological susceptibility of the most-sensitive pre-adult stage—which is based on physiological differences between the most sensitive juvenile state and adults of the species—can be complex. In species where the juvenile is likely to be more susceptible than the adult, the species overall sensitivity will be greater than in species where the juvenile stage is less susceptible than adults (McIntosh et al. 2015). For instance, turtle hatchlings are more susceptible to entanglement than adults when they try to reach the sea, because they are less able to break free from entanglements and are highly vulnerable to predation at this stage (Triessnig et al. 2012). In some species, juvenile birds are at higher risk from ingestion and entanglement than adult birds, because of regurgitative feeding and increased time in nests, respectively (Raum-Suryan et al. 2009). In other taxa, if juveniles are too small to consume macroplastics or become entangled, as is the case for many fishes and invertebrates, adults will be the most sensitive life-stage (Nunes et al. 2018). Overall, intraspecies variation in entanglement and ingestion sensitivity across life stages was documented in both marine vertebrates and invertebrates, but the directional of sensitivity was species dependent.

Finally, marine organisms are not exposed to macroplastic pollution in isolation from other environmental stressors. Reduced fitness due to other has been associated with increased interactions with macroplastic and detrimental consequences of these interactions (Drever et al. 2018, LaCombe et al. 2020). Climate change and other stressors can reduce food availability, driving animals to broaden their diet and consume more plastics. For example, a mortality event of Red Phalaropes was linked to reduced upwelling—an important food source—due to unseasonably warm ocean temperatures. All carcasses were severely underweight and 100% contained plastics (Drever et al. 2018). In odontocetes, parental loss and central nervous system disease were also both identified as risk factors for plastic ingestion (LaCombe et al. 2020). Environmental stressors, such as pollution, climate change and increased human activity, can increase the likelihood of disease and death of a mother leading to mother-calf separation (Fair and Becker 2000). Overall, many studies identified relationships between plastic pollution sensitivity and exposure to other stressors, including disease, climate change, nutrient pollution, vessel strikes. These relationships were documented in primarily in marine vertebrates, but also mentioned for marine plants and corals (Lamb et al. 2018, Suyadi & Manullang 2020).

Traits associated with population resilience. Six traits influence a species' resilience to population decline due to cumulative individual mortalities or reduced fitness from exposure to plastic ingestion or entanglement—abundance, population connectivity, reproductive turnover, behavioral specialization, sensitivity of most important life stage, and risk of extinction (Table 9). Four of these traits (abundance, population connectivity, reproductive turnover, and feeding or habitat specialization) were also employed by Polidoro et al. (2021).

Table 9

Population Resilience Component of the Macroplastic Vulnerability Index Framework

Population resilience						
Trait	Abundance	Population connectivity	Reproductive turnover rate	Feeding or habitat specialization	Importance of most impacted life stage	Species extinction risk
Assumption	Populations with fewer individuals will be less resilient	Populations with little or no connectivity to populations outside high-risk zones will be less resilient	Species with lower turnover rates will recover more slowly	Species with high specialization in habitat and/or dietary choice are less resilient	Species where the most sensitive life-stage is of high importance for population maintenance are less resilient	Species with higher risk of extinction are less resilient
Example indicators	•Population size	•Connectivity with populations in or outside of high impact areas	•Offspring per year •Generation length •Recruitment rate	•Number of habitat preferences •Number of food preferences	•Population importance of the most sensitive life-stage	• IUCN status

If population size is small, the loss of individuals due to environmental stressors is more likely to cause local extinction than if the population size is large (Dulvy et al. 2003). Population connectivity similarly influences the resilience of local populations. If a vulnerable population has high connectivity with resilient populations of the species, then immigration can reduce local extinction risk and increase resilience, but if connectivity to resilient populations is low then local extinction risks increase (Jones et al. 2007). Importantly, connectivity to maladaptive populations may reduce population resilience (McManus et al. 2021).

Reproductive turnover may influence a population's resilience to disturbance, including plastic pollution. Slower reproductive turnover (i.e., K strategists) is associated with a higher sensitivity to stressors than species with high reproductive turnover (Dulvy et al. 2003). Reproductive turnover can be measured as generation time, number of offspring, reproductive age, and population turnover rate (Polidoro et al. 2021, Mace et al. 2008). Though population level studies are limited, one study found plastic ingestion in albatrosses likely led to population decline because they are long-lived species with slow reproductive turnover (Roman et al. 2021). Entanglement of South American fur seals, another species with a slow reproductive turnover, was also linked to population decline (Perez-Venegas et al. 2021). Finally, less specialized species are generally more resilient to local and regional stressors, as they can adapt their behaviors, habitats and feeding preferences more readily than species that are highly specialized (Ducatez et al. 2020).

Available data suggest high intraspecies variation in vulnerability to plastic pollution among life stages. For instance, juveniles are often more vulnerable to entanglement than adults (Page et al. 2004, Dau et al. 2009, Costa et al. 2020). Certain life stages are more important for population maintenance than others (Gerber and Heppell 2004), and this is often species dependent (e.g., adult males, immature females, new borns, juveniles). If the most important life stage is also the most sensitive to plastic pollution than population resilience will be disproportionately low. For example, even small amounts of entanglement of adult female South American fur seals had large population effects, because of the subsequent decrease in the number of offspring the colony produced (Perez-Venegas et al. 2021). Finally, species populations that are already at risk of extinction are less resilient to new stressors. In such cases, plastic pollution can directly influence extinction risk for threatened and endangered species (Good et al. 2010). For example, entanglement-induced injury and death from marine debris in the Hawaiian Islands has hindered recovery efforts for the endangered Hawaiian Monk Seal (*Monachus schauinslandi*) (Boland and Donohue 2003).

Framework application.

The resulting framework includes a comprehensive list of biological, physiological, and ecological traits identified in my literature review that influence the vulnerability of marine species to macroplastic. Users can apply this framework, following the seven steps described below, to develop vulnerability indices that estimate the relative vulnerability of marine species to macroplastic (Figure 3). These indices can then be used to identify populations or ecosystems for long-term monitoring or to inform policy and management priorities.

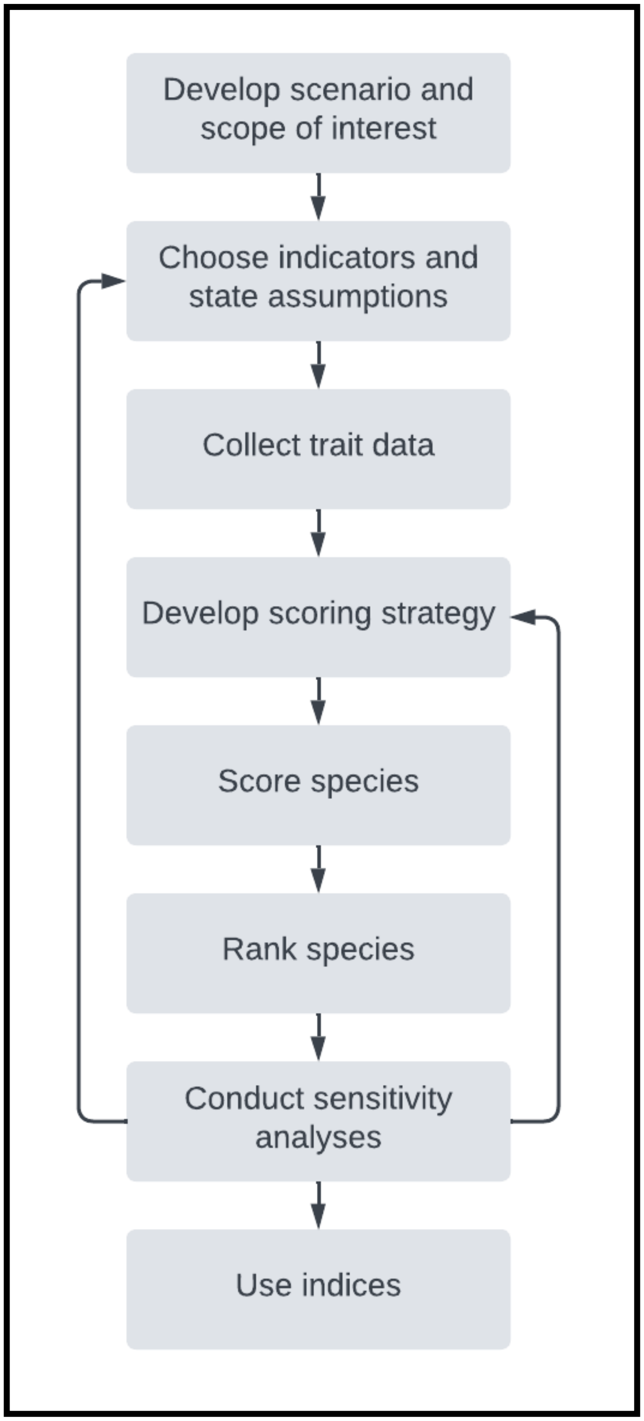

Figure 3. Steps for framework implementation to develop a vulnerability index

Identify scenario of interest. The scenario should be informed by the management or research objectives of the implementer. This should include the species of interest (e.g., marine mammals, species of economic importance), the region of focus (e.g., local, global), the types of plastic pollution (e.g., fishing nets), and possibly a focus on ingestion vs. entanglement. Plastics represent a category of various pollutants that interact with the environment and species differently. For example, if the goal of the manager is to prioritize marine regions for marine debris removal and long-term monitoring in the Northwest Hawaiian Islands (NWHI), the scope may be local—the NWHI—include all species native to this region, and focus on fisheries-based marine debris, as this is the primary plastic pollutant in this isolated group of uninhabited islands. Fishing-related plastics can have different positions in the water column or create a higher likelihood of mortality from entanglement than consumer plastics, such as plastic bags. Moreover, geographical context can be critical to inform interactive effects from another local anthropogenic stressor. For example, both coral bleaching events and plastic pollution can increase risk of coral disease, and these effects are likely additive (Lamb et al. 2018). Therefore, a clearly defined scenario that considers the context for which the vulnerability index is being develop is important.

Choose indicators & state assumptions. My literature review included all taxa, so not every trait in my framework will be relevant for every scenario. For instance, though respiratory mode affects vulnerability, a within taxa analysis of mammals would not include respiratory mode as it is it would not inform relative vulnerability. Indicators for each included trait should also be scenario-specific and informed by both the traits that are most important for distinguishing the vulnerability of the study species and data availability. For example, an evaluation focused on seabirds should include a non-foraging behavior indicator related to nesting habits, while curiosity or aggression would be more appropriate indicators of non-foraging behaviors influencing marine mammal sensitivity. While choosing the best indicators, it is also important to clearly state the associated assumptions (e.g., longer life expectancy increases likelihood of exposure). These assumptions will depend on the target species. For instance, decreasing size may reduce sensitivity to entanglement in fishes if they are too small to become entrapped in marine plastics. Alternatively, increased size may reduce sensitivity to entanglement in marine mammals by making it easier for them to break free from entanglements. If the target species were only mammals, then the assumption may be that increasing size will decrease sensitivity, if the target species are only fish, the assumption may be decreasing size will decrease sensitivity and if both are included the of size on sensitivity may be bimodal.

Collect trait data. The next step is to compile available biological and ecological trait data. Sources outside of peer-reviewed, academic literature, such as the IUCN database, are important sources as well.

Develop scoring strategy. Scoring metrics require consideration of how to distinguish species of interest for each indicator. For instance, an assumption may be that long-lived species have higher exposure than a short-lived species. Longevity scores could be classified in scored categories between 1 and 5, with five having the longest average life expectancies, with set cut offs or thresholds between each integer score. It is important to consider how data gaps will be scored (e.g., De Lange et al. 2009; Golden and Rattner

2003). Unknown traits are often given a score of 3 following a precautionary approach (Woodyard et al. 2022).

Score, rank and categorize species. Based on available trait data, each species should be scored using the indicators and scoring metrics in place, with equal weight being put on each vulnerability dimension—likelihood of exposure, sensitivity, and population resilience—in the final score. The species of interest can be ranked in order from most to least sensitive by their scores. It is important to note that the difference in scores cannot be used to quantify differences in vulnerability (i.e., a score of 90 vs 45 does not mean one species is 2x more vulnerable). Instead, it provides information on the relative sensitivity of two species (i.e., which is more vulnerable). It is good practice to categorize final scores into categories of vulnerability rather than focusing on absolute scores. For example, species with scores in the top quartile might be classified as having high vulnerability, while species with scores in the bottom quartile are classified as having low vulnerability.

Conduct sensitivity analyses. After the first round of scoring, ranking and categorization is complete, it is important to validate rankings and conduct sensitivity analyses to ensure 1) indicators meaningfully contribute to the rankings, and 2) the scoring strategy properly weights traits. This can be done by removing or changing indicators, reranking species and validating rankings using species with more data in the literature or expert elicitation. If an indicator does not contribute to the ranking—due to significant data gaps (e.g., little is known about habitat use by species of interest) or negligible variation in the indicator among species (e.g., respiration mode would be the same for all marine mammals)—it should be removed, species should be rescored, and the new ranking should be validated. This should also be done if two traits representing the same vulnerability dimension have statistically significant correlation. Finally, if relative rankings still do not reflect existing data (e.g., a species with high documented rates of entanglement) ranks low in species sensitivity), and the literature suggests certain traits have greater importance, then weighting of specific traits may be used to improve ranking accuracy. Other studies have done this by giving certain traits a multiplier that increase its relative importance to other traits in the final scoring (Golden and Rattner, 2003).

Use indices. After the final ranking is determined, it can be used to inform future research and decision-making. For instance, communities and marine regions of high vulnerability can be mapped and identified based on species' distributions (e.g., Foden et al. 2013, Davidson et al. 2012). This can inform regions to prioritize for mitigation efforts or long-term monitoring.

Conclusion

Marine plastic pollution is ubiquitous in our global oceans. Despite evidence that marine biota is impacted by macroplastics, little is known about the impact of plastic at the population, community, or ecosystem level. Managers and policymakers need risk assessment frameworks to inform and prioritize conservation action. My comprehensive trait-based framework aims to help researchers and decision-makers use existing data to evaluate the relative vulnerability of populations and communities to marine macroplastics, within or across taxa and marine regions, at any spatial scale or geography.

The impacts of plastic pollution on populations, species, and communities are confounded by other anthropogenic stressors facing marine biota; however, my review indicated limited research has been done to understand the impacts of multiple stressors on marine biota's vulnerability to plastic pollution. My framework can be used to identify vulnerable populations, species, and ecosystems that should receive targeted management and mitigation efforts, as well as long-term monitoring of population and community health. Long-term monitoring of vulnerable marine biota would provide insight into the efficacy of mitigation efforts and could provide key locales for studying how exposure to multiple stressors are affecting species vulnerable to plastics. TBAs represent a relevant tool to inform regulatory frameworks for ecological risk assessments on macroplastics. The negative consequences of macroplastic pollution are evident, yet the lack of ecologically informed limits for plastic pollution make regulatory management difficult. Implementation of TBAs for marine species in ecological risk assessment frameworks can facilitate identification of data gaps and effective regulatory action.

CHAPTER 5

A MACROPLASTIC VULNERABILITY INDEX FOR MARINE MAMMALS, BIRDS, AND TURTLES IN HAWAIʻI

Introduction

Plastic pollution is ubiquitous in our global oceans with increasing impacts on marine organisms (Bucci et al., 2020; Borrelle et al., 2020). To date, interactions with marine plastic pollution have been documented in more than 1300 marine species (Kühn and Van Franeker, 2020; Santos et al. 2021). These interactions are both physical and chemical and vary depending on plastic size (Thornton-Hampton et al., 2022; Bucci et al., 2020). For macroplastics (>5 mm in diameter), physical interactions pose the greatest documented threat to marine biodiversity (Bucci et al. 2020). Understanding impacts of plastic pollution at the higher levels of biological organization is critical to addressing and mitigating ecological consequences (Bucci et al. 2020, Koelmans et al. 2017). In this paper, I apply a traits-based approach (Murphy et al. in review) to assess the vulnerability of Hawaiian marine species to macroplastic pollution and exemplify the potential of this approach for plastic pollution research and management across the globe.

Physical exposure to macroplastic pollution occurs primarily via ingestion and entanglement, which have been observed across a wide range of marine taxa (Kühn and Van Franeker 2020, Santos et al. 2021; Jepsen et al. 2019, Ryan et al. 2018, Lamb et al. 2018). Most research has been conducted on organismal exposure to macroplastics through these pathways, with research on effects—such as injury, illness, or mortality—focusing primarily on the sub-organismal and organismal level (Bucci et al. 2020). As a result, very little is known about the physical impacts of macroplastics at the population, assemblage, or ecosystem levels (Murphy et al. *in review)*.

A promising method for cross-organization inference is the use of trait-based approaches (TBAs) (Koelmans et al. 2017). TBAs use information on the ecological, physiological, and biological traits that influence organisms' vulnerability to a stressor to predict the relative vulnerability of understudied species based on their traits. TBAs are robust and can improve ecological risk assessments when data are limited by allowing for extrapolation between levels of biological organization, and across spatial and temporal scales (Van den Brink et al. 2011). Such analyses generally involve the development of vulnerability indices (i.e., rankings of species' relative vulnerability to a stressor) to understand a given stressor's population and assemblage level impacts. Such indices have already been implemented to inform research and management of several other anthropogenic stressors, including pesticides, metals, pharmaceuticals, lead shot, oil, and climate change (Polidoro et al. 2021; Foden et al. 2013, Chin et al. 2010, De Lange et al. 2009, Golden and Rattner 2003).

The potential value of TBAs for plastic pollution has been exemplified through a few studies, though their application has been limited in scope (Good et al. 2020; Compa et al. 2019). To facilitate more consistent and broader applications of TBAs for plastic pollution, in the previous chapter, I present a multi-taxonomic approach for developing macroplastic vulnerability indices. Through a comprehensive literature review, I identified 22 traits that have been shown to influence species vulnerability to plastic pollution along three dimensions: likelihood of exposure, species' sensitivity, and population resilience.

This work provides steps to apply the resulting framework to develop a vulnerability index for any marine species or geographic area.

Here, I apply the framework presented in the previous chapter to develop a multi-taxonomic vulnerability index for marine mammals, seabirds, and sea turtles in the Hawaiʻian exclusive economic zone (EEZ). Hawaiʻi presents a valuable case study given the well-documented high densities of marine plastic pollution, vulnerable marine species, and evidence of organismal interactions with macroplastic (Hawaii DLNR, 2022; NOAA Marine Debris Program, 2021). I develop an index that provides insight into the Hawaiian species that are most and least vulnerable to macroplastic pollution. I discuss the generality and limitations associated with the broad application of my method.

Methods

To develop my relative multi-taxonomic vulnerability index, I followed the steps outlined in the previous chapter - (1) identify the scope of interest, (2) choose indicators and state assumptions, (3) collect trait data, (4) develop scoring strategy, (5) score and rank species, (6) conduct sensitivity analyses.

Identify the scope of interest. I focused on three taxa—marine mammals, sea birds, and sea turtles—in the Hawaiian EEZ. Hawaiʻi is biodiverse, with the highest proportion of endemism of any tropical marine ecosystem on Earth (Fautin et al., 2010). It is also known as the endangered species capital of the world (Hawaii DLNR, 2022). While comprising less than one percent of the United States land mass, Hawaiʻi contains 44 percent of the nation’s Endangered and Threatened plant species (USFWS, 2022), and plastic pollution has been identified as a potential threat for many of these species (IUCN, 2022). Most of the marine plastic pollution in Hawaiʻi comes from external sources, which make clean-up and remediation an important part of a local marine debris management plan (NOAA Marine Debris Program, 2021). Therefore, government officials, non-governmental organizations and other groups managing plastic pollution and conserving marine species in Hawai‘i would benefit from a relative vulnerability index to inform priorities and identify the best species to monitor for population decline.

I applied the trait-based approach to all taxonomically valid marine mammals (25 species), seabirds (33 species), and sea turtles (5 species) present in Hawaiian waters, because the physical exposures of macroplastic pollution are well-documented in these taxa (Kühn and Van Franeker, 2020; Bucci et al. 2020). Additionally, these taxa hold social and ecological importance to marine systems globally (Tavares et al. 2019). Importantly, by including three taxonomically distinct groups, I exemplify the functionality of the framework for multi-taxonomic analyses. Finally, I chose to focus on the physical vulnerability of species to macroplastics due to ingestion and entanglement, as all three taxa are influenced by both types of interactions (Kühn and Van Franeker, 2020; Senko et al. 2020).

Choose traits, choose indicators, and state assumptions. A first step in applying the framework is selecting relevant traits. I included 11 of the 22 traits presented in the multi-taxonomic vulnerability framework: two linked to likelihood of exposure to macroplastics, five to species’ sensitivity to ingestion and/or entanglement, and four to overall population resilience (Table 1). Trait selection was based on data availability as well as their usefulness for distinguishing the study species.

Table 10

Traits from the Multi-Taxonomic Vulnerability Framework with Traits Included in this Analysis Highlighted

Likelihood of Exposure	Species Sensitivity	Population Resilience
Distribution	Body morphology	Extinction Risk
Longevity	Feeding/foraging behavior	Population size
Motility	Prey preferences	Reproductive turnover rate
Longevity of pre-adult	Reduced fitness from other stressors	Habitat specialization
Distribution of pre-adult	Egestion potential	Population connectivity
Habitat	Pre-adult behavior	Importance of most sensitive life stage
Water column position	Non-foraging behavior	
	Respiration mode	
	Physiological sensitivity of pre-adults	

The two traits included for likelihood of exposure are distribution and longevity. I used average density of surficial macroplastic pollution (from Eriksen et al. 2014) within the species' range as an indicator for distribution (quantified using species range data and plastic distribution maps), based on the assumption that the higher the density of macroplastic in a species range (items per km^2) the more likely an individual is to encounter it (See Appendix B for methods to quantify items per km^2). Expected maximum life span was the chosen indicator for longevity, assuming longer-lived species have more opportunities for plastic interactions.

Motility, habitat, longevity of the most sensitive pre-adult stage, distribution of the most sensitive pre-adult stage, and water column position were excluded due to data availability. For example, some habitats have been linked with plastic capture and accumulation (e.g., mangroves), but research is not available on plastic accumulation rates or taxa use for all habitat types (Luo et al. 2021).

The five traits included for species' sensitivity were body morphology, feeding and foraging behavior, prey preferences, egestion potential, and vulnerability to other stressors. Body mass was the indicator chosen for body morphology, assuming that species with higher body mass are less sensitive to drowning if entangled and are less sensitive to negative impacts from ingestion (Kaplan Dau et al. 2009, Thiel et al. 2018). Sensitivity associated with different foraging behaviors and prey preferences were informed by the literature (Roman et al. 2019, Thiel et al. 2018, Bond et al. 2013). I used regurgitation potential as an indicator of egestion potential, as species that can regurgitate indigestible plastics more easily are less sensitive to ingestion (Basto et al. 2019). Finally, I used listed threats from each species assessment on the IUCN Red List of Threatened Species (www.iucnredlist.org) as an indicator for vulnerability to other stressors. Species experiencing significant impacts from other stressors are likely more sensitive to macroplastic pollution; therefore, I assumed species experiencing more threats are more likely to experience other stressors that compound plastic pollution (Drever et al. 2018, LaCombe et al. 2020). I excluded respiratory mode because all species selected have the same mode of respiration. Non-foraging behavior, pre-adult behavior and relative sensitivity of pre-adult stages were excluded due to data availability.

Four out of six traits were included to inform population resilience: abundance, specialization, reproductive turnover rate, and risk of extinction. Population size was used as the indicator for abundance, as smaller populations are less resilient (Dulvy et al. 2003, Mace et al. 2008). I chose number of habitats as the indicator for specialization, assuming species that are more specialized are less resilient (Ducatez et al. 2020). Generation length, defined as the average age of reproducing adults, was selected as the indicator for reproductive turnover rate, as species with longer generation lengths have populations that recover more slowly from disturbances (Dulvy et al. 2003). Finally, I used IUCN Red List status as an indicator of extinction risk. I excluded population connectivity, because it is difficult to identify the role of connectivity in improving population resilience for large ranged species with complex migration patterns (McManus et al. 2021; Compa et al. 2019). I excluded the relative importance of the most sensitive life stage due to limited knowledge of population structure and intra-life stage variation in species' sensitivity for most species.

Collect trait data. To collect species specific trait data, I used a variety of databases and organizations, including the IUCN Red List, Birds of the World, Animal Diversity Web, National Oceanic and Atmospheric Administration, and Sea Turtle Conservancy (IUCN 2022, Birds of the World 2022, Myers et al. 2022, NOAA 2022, Sea Turtle Conservancy 2022). I then addressed data gaps using peer-reviewed literature. Macroplastic concentration maps were taken from Eriksen et al. 2014.

For continuous, quantitative indicators—plastic density per km^2, longevity, mass, population, and average generation length—I used quantitative data whenever available and converted data provided to a mean value with a standard deviation (SD). When the data source provided a single value with high confidence (e.g., population size), I assumed this was the mean value for the species with no SD. When a single value was provided with a statement of uncertainty (e.g., approximate population size), I assumed the provided value was the mean, but included a 10% standard deviation to be conservative. When a range was provided by a single data source or two sources provided conflicting values (e.g., population is 100,000 to 300,000), I assumed the range given had a 95% confidence interval. In this instance, I used the average of the two values as the mean (e.g., 200,000), and assumed the range captured two SDs in each direction (e.g., SD is 50,000). If quantitative data were not available, I included qualitative information provided. For example, the population size of Pygmy Sperm Whale (k. Breviceps) is unknown, but is considered a rare species (IUCN, 2022). Therefore, general abundance was coded as “rare”. If data were not available for a certain species, I assigned a best estimate based on data available for other species (e.g., used American Coot mass for the Hawaiian Coot), and included an SD of 10% (See supplementary materials for more detail).

For non-continuous or categorical indicators—feeding behavior, prey preferences, listed IUCN threats, regurgitation potential, IUCN Red List status, and habitat specialization—I collected all available information from the provided databases. For number of habitats and the number of IUCN threats each species was exposed to, I summed the number listed in the IUCN database (IUCN, 2022), and assumed no standard deviation (because no uncertainty was provided). For other traits, I recorded qualitative data (See Appendix B for more detailed methods on indicator calculations and Appendix C for trait data)

Develop scoring metrics. All indicators were scored on a scale of one to five, with one representing the lowest possible contribution to vulnerability and five being the highest to ensure all traits were equally weighted within a given vulnerability dimension (e.g., distribution and longevity had equal influence on likelihood of exposure scores). Table 11 provides a summary of the scoring metrics used for each indicator. For the continuous quantitative traits, I calculated quintiles to identify the cut-off points for scores. For unknown population sizes with qualitative descriptors, “rare” species were scored a five, “fairly common” species were scored a two and “unknown” species were scored a three, all with a SD of 1 for the score. For categorical data, I developed scores based on the stated assumptions and the Chapter four literature review. For example, each IUCN Red List status corresponded to a number from one to five with least concern species receiving a score of one and critically endangered species receiving a score of five. For unknown categorical data a score of three was used with an SD of one for the score (See Appendix B for more detailed information on scoring).

Regurgitation potential, feed and foraging behavior, and prey preference score categories were informed by the literature (Good et al. 2020; Roman et al. 2022; Andrades et al. 2019; Ryan et al. 2019; Schuyler et al. 2014). Importantly, both the traits of species and the traits of the plastic influence the likelihood of ingestion and entanglement. For example, surface seizing birds eat more hard fragments on the surface, while turtles consume more films. This is because plastic traits influence their occurrence in the water column and how similarly they resemble prey items (Ryan et al. 2019; Schuyler et al. 2014).

Table 11

Traits, Indicators, and Ranking Assumptions for Species Scoring

Vulnerability dimension	Trait	Indicator	Ranking Assumptions
Likelihood of exposure	Distribution	Average density of plastic in species range	Quintiles (Low = 1 to High = 5)
	Longevity	Life span	Quintiles (Low = 1 to High = 5)
Species' sensitivity	Body morphology	Body mass	Quintiles (High = 1 to Low = 5)
	Feeding and foraging behaviors	Foraging behavior influence on ingestion rate	1 = pick and probe; under water pursuit; pursuit diving; stealing food in flight; chase prey 2 = biter; plunge diving 3 = dabbling; swallower; deep dive 4 = fluttering on surface; dipping; grazer 5 = surface seizing; scavenging; filter feeding

	Prey preferences	Interaction risk based on prey type	1 = Specialist that does not eat prey resembling plastic, feed on waste, or feed over fisheries species 2 = Generalist that does not eat high risk prey 3 = Generalist that eats some high risk prey 4 = Specializes on prey sometimes mistaken for plastic or feeds over fisheries species 5 = Specializes on prey commonly mistaken for plastic or fisheries species, or feeds on human waste
	Reduced fitness from other stressors	IUCN threat list	Score calculated based on number and severity of threats. Each stressor had a severity score (1-8) and these severity scores were summed for all stressors to give a total threat score.
	Egestion potential	Ability to regurgitate or use of gastroliths	1 = Regurgitate pellets frequently and regurgitate to young 2 = Regurgitate and limited pellet production observed; occasional pellet casting in young; may produce pellets based on species 3 = Capable of regurgitation or ingest gastroliths 4 = May regurgitate to young; may regurgitate based on species 5 = Does not regurgitate to offspring and no evidence of pellets; anatomical structure reducing regurgitation potential; no information
Population resilience	Abundance	Population size	Quintiles (High = 1 to Low = 5)
	Specialization	Habitat number	Quintiles (High = 1 to Low = 5)
	Reproductive turnover rate	Generation length	Quintiles (Low = 1 to High = 5)
	Extinction risk	IUCN Red list status	1 = Least concern 2 = Near threatened 3 = Vulnerable 4 = Endangered 5 = Critically endangered

Calculation of species scores and final rankings. Each species received a score for every trait based on the scoring system developed (Table 2). Trait-specific scores were then put into Equation 2 to calculate a final relative vulnerability score for every species.

$$\text{Vulnerability score} = ((\Sigma T_{1\text{-}2})/2 + (\Sigma T_{3\text{-}7})/5 + (\Sigma T_{8\text{-}11})/4)/3 * 20 \quad (2)$$

T_1 and T_2 represent the two likelihood of exposure traits—distribution and longevity—T_3 to T_7 represent the five species' sensitivity traits and T_8 to T_{11} represent the four population resilience traits. Therefore, the equation weighs each dimension of vulnerability—likelihood of exposure, species' sensitivity, and population resilience—equally by finding a mean score out of five for each dimension. These three scores are then averaged and multiplied by 20 so each species has a possible total vulnerability score between 20 and 100. I then identified quintiles for the total vulnerability score and each species was placed into one of five vulnerability groups: low (20-45.44), low-medium vulnerability (45.44-58.58), medium vulnerability (58.58-64.52), medium-high vulnerability (64.52-69.88), or high vulnerability (69.88-100).

When calculating species final scores, I used bootstrapping in my analyses to account for uncertainty in trait data. For each species' trait value, I generated 1,000 random values, assuming a normal distribution around the recorded mean and SD. This assessment included 63 species, ultimately producing 63,000 total estimates for a single trait for all species. I then used these values to develop the quintile cut offs for all continuous, quantitative traits (i.e., quintile cut-offs were based on 63,000 generated values based on SD within data, instead of based on mean values alone). I applied the quintile cutoff points to all 1,000 trait estimates for each species to generate 1,000 scores for a given trait. Finally, I used Equation 2 to calculate 1,000 final vulnerability scores for each species. From these 1,000, I calculated the mean vulnerability score and identified the standard error (two SD) for each species score. All analyses in Rstudio Version 2022.02.2+485 "Prairie Trillium" Release (See Appendices C and E for detailed description of methods and R script, respectively).

It is important to note that a high standard error for trait data did not always lead to a high error in score. This is because if the range provided all fell within one quintile, then the score for a given trait would still always be the same. For example, a population range could be 10,000,000 to 15,000,000, but even the lowest population size in this range is still high enough to produce a score of one for population.

Sensitivity Analyses. To ensure that all the traits included in my analysis were important in determining species scores, I conducted sensitivity analyses. I first calculated correlation between traits and tested for significant correlation. I then recalculated vulnerability scores, removing traits that were correlated with another trait in the same vulnerability dimension (e.g., removed generation length due to correlation with population abundance) to ensure no traits were redundant. Ultimately, no traits were redundant, and all 11 traits were included in the final analysis. To identify the sensitivity of results to trait data quality, I recalculated vulnerability scores by increasing all SD values of zero to 1.25 and 2.5 (2.5% SE and 5% SE). I then identified how confidence in trait data values influenced confidence in the final vulnerability groups (Results in Appendix B).

Results

Figure 4 shows the final vulnerability scores, as well as scores for each dimension, by taxonomic group. On a scale from 20 – 100, final species scores ranged from 33 to 82.9, indicating a wide range of vulnerability. Based on quintiles of final relative vulnerability scores, thirteen species were categorized as low vulnerability, 13 as low-medium, 12 as medium, 13 as medium-high, and 13 as high (Table 3). Generally, differences in vulnerability can be seen by taxonomic group (Figures 4 and 5). All 13 low vulnerability species are birds, primarily ducks (*Anatidae*), and waders (*Rallidae, Scolopacidae, Charadriidae,* and *Ardeidae*). This group also includes three noddies (*Larridae*), the white-tailed tropic bird (*p. Lepturus)* and the grey-backed tern *(o. Lunatus).* Species in the lowest vulnerability group typically had scores for exposure and population resilience in the lowest quintile; however, they varied in their sensitivity, with the Blue Gray Noddy falling into the highest quintile for its sensitivity scores due to its prey preferences, feeding behaviors and regurgitation potential (Table 12).

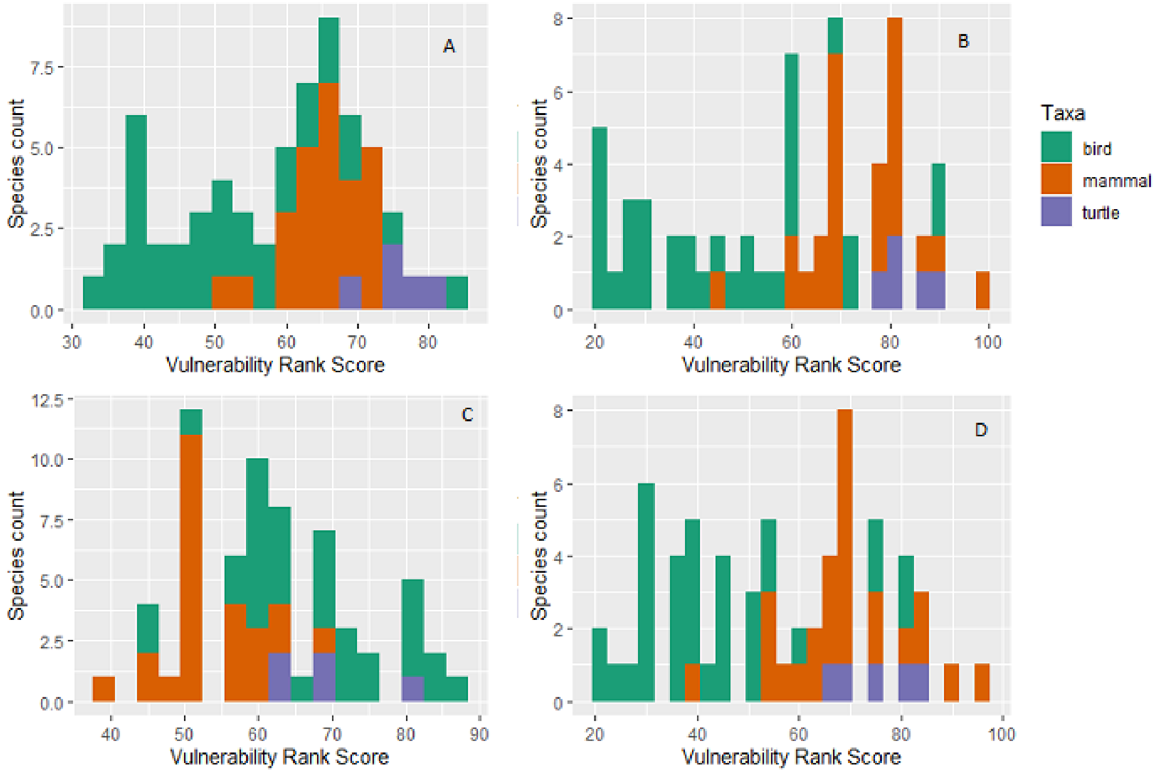

Figure 4. Vulnerability Scores by Taxa. (A) Distribution of total scores by taxa. (B) Distribution of exposure scores by taxa. (C) Distribution of sensitivity scores by taxa. (D) Distribution of resilience scores by taxa.

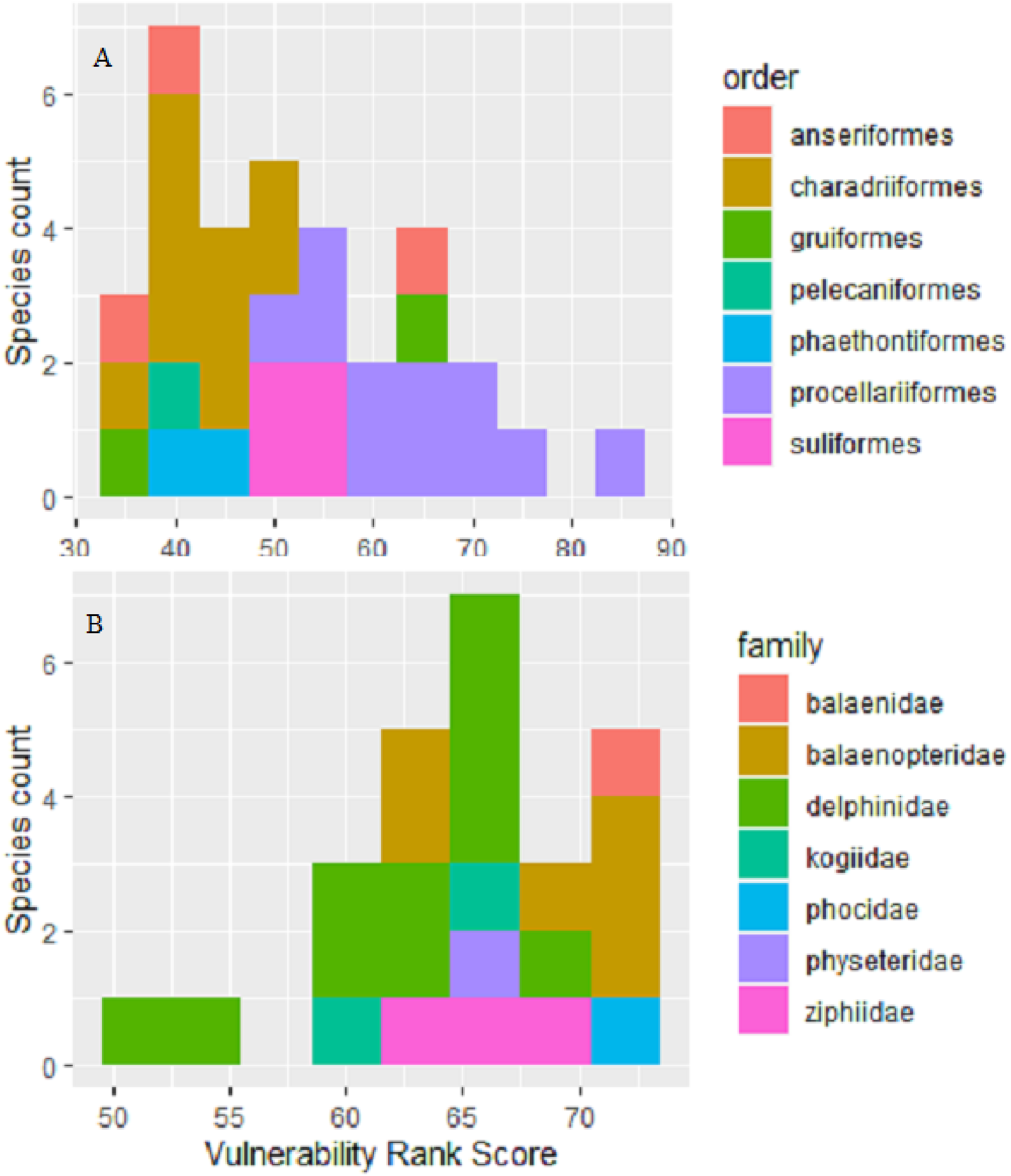

Figure 5. Vulnerability Scores by Species Group within Taxa. (A) Distribution of total scores for mammals by order. (B) Distribution of final scores for birds based on family.

Table 12

Vulnerability Index for Hawai‘ian Marine Mammals, Turtles, and Birds*

Species	Vulnerability Group	Vulnerability Rank	Exposure	Species sensitivity	Population resilience	Taxa
Sanderling (*c. Alba*)	Low	33	20	44	35	Bird
American Wigeon (*m. Americana*)		35.2	26.1	59.6	20	Bird
Eurasian Moorhen (*g. Chloropus*)		36.8	20	70.4	20	Bird
Ruddy turnstone (*a. Interpres*)		38.1	23.4	63.9	27	Bird
Northern Shoveler (*s. Clypeata*)		39.2	20	67.6	30	Bird
Wandering Tattler (*t. Incana*)		39.7	20	44	55	Bird
Black-crowned Night Heron (*n. Nycticorax)*		39.9	26.3	62.5	31.1	Bird
Pacific Golden Plover (*p. Fulva*)		40	26.2	58.8	35	Bird
White-tailed Tropicbird (*p. Lepturus*)		40	30.1	60	30	Bird
Grey-backed Tern (*o. Lunatus*)		40.5	30.3	60	31.3	Bird
Blue Gray Noddy (*a. Ceruleus*)		43.5	21	74.5	35	Bird
Black Noddy (*a. Minutus*)		43.7	29.1	72	30	Bird
Brown Noddy (*a. Stolidus*)		45.4	39.8	66.1	30.5	Bird
Red-tailed Tropicbird (*p. Rubricauda*)	Medium-Low	47.1	44.3	52	45	Bird
Masked Booby (*s. dactylatra*)		48.5	41.6	64	40	Bird
Red Footed Booby (*s. Sula*)		49	50.9	56.2	40	Bird
Christmas Shearwater (*p. Nativitatis*)		49.6	34.9	64	50	Bird
Sooty Tern (*o. Fuscatus*)		51.3	60.1	68.8	25	Bird
Fraser’s Dolphin (*l. hosei*)		51.4	45.2	56	53.2	Mammal
White Tern (*g. alba*)		52.2	35.9	80	40.8	Bird
Brown Booby (*s_leucogaster*)		54.3	58	60.1	45	Bird
Great Frigate Bird (*f_minor*)		55	60	60	45	Bird
Spinner Dolphin (*s. Longirostris*)		55.3	70	56	40	Mammal
Bonin Petrel (*p. Hypoleuca*)		56	72.1	56	40	Bird
Wedge-tailed Shearwater (*a. Pacifica*)		56.7	55	80	35	Bird
Band-rumped Storm Petrel (*h. Castro*)		58.6	49.7	76	50	Bird

Bulwer's Petrel (*b. Bulwerii*)	Medium	59.1	49.4	88	40	Bird
Pygmy Sperm Whale (*k. Breviceps*)		60.3	60	56	65	Mammal
Pygmy Killer Whale (*f. Attenuata*)		60.5	60	52.4	69.3	Mammal
Risso's Dolphin (*g. Griseus*)		60.6	64.9	60	56.8	Mammal
Pantropical Spotted Dolphin (*s. Attenuata*)		62.1	80	51.2	55	Mammal
Common Minke Whale (b. *Acutorostrata*)		62.3	70	52	65	Mammal
Newell's shearwater (*p. Newelli*)		62.5	39.6	68	79.9	Bird
Melon headed Whale (*p. Electra*)		62.8	66.5	52	70	Mammal
Tropical Bottlenose Whale (*i. Pacificus*)		63.9	61.4	56	74.3	Mammal
Hawaiian Coot (*f. Alai*)		64.0	60	71.9	60	Bird
Humpback Whale (*m. Novaeangliae*)		64	70	52	70	Mammal
Striped Dolphin (*s. Coeruleoalba*)		64.5	98.4	40	55	Mammal
Rough-toothed Dolphin (*s. Bredanensis*)	Medium-High	64.8	78.6	50.6	65	Mammal
Short-finned Pilot Whale (*g. Macrorhynchus*)		65.1	86.9	48.3	60	Mammal
Laysan Duck (*a. Laysanensis*)		65.3	60	60.9	75	Bird
Dwarf Sperm Whale (*k. Sima*)		65.8	69.8	64.1	63.7	Mammal
False Killer Whale (*p. Crassidens*)		66	77.6	45.3	75	Mammal
Tristram's Storm Petrel (*h. Tristrami*)		66.3	60	84	55	Bird
Cuvier's Beaked Whale (*z. Cavirostris*)		66.8	70	60	70.4	Mammal
Sperm Whale (*p. Macrocephalus*)		67.3	80	52	70	Mammal
Laysan Albatross (*p. Immutabils*)		68.6	88.9	72	45	Bird
Black-footed Albatross (*p. Nigripes*)		68.9	72.8	84	50	Bird
Olive Ridley (*l. Olivacea*)		69	77.9	64	65	Turtle
Blainville's Beaked Whale (m. *Densirostris*)		69.5	80	64	64.4	Mammal
Orca (*o. Orca*)		69.8	77.4	52	80.1	Mammal
Bryde's Whale (*b. Edeni*)	High	70.4	89.3	52	70	Mammal
North Pacific Right Whale (e. *Japonica)*		70.7	70	52	90	Mammal
Sei Whale (*b. Borealis*)		71.7	70	60	85	Mammal

Fin Whale (*b. Physalus*)	72.3	80	52	85	Mammal
Hawaiian Monk Seal (*n. Schauinslandi*)	72.6	79.8	68	70	Mammal
Blue Whale (*b. Musculus*)	73	80	44	95	Mammal
Green sea turtle (*c. Mydas*)	74.7	80	68.8	75.4	Turtle
Hawaiian Petrel (*p. Sandwichensis*)	75	70.1	80	75	Bird
Loggerhead (*c. Caretta*)	76.1	90	68.2	70	Turtle
Hawksbill (*e. Imbricata*)	77.4	88.4	63.9	80	Turtle
Leatherback (*d. Coriacea*)	81.4	80	80	84.3	Turtle
Short-tailed albatross (*p. Albatrus*)	82.9	88.7	80	80	Bird

*Green signifies the lowest quintile species within a vulnerability dimension or total vulnerability. Yellow signifies the second lowest, orange the middle, red the second highest and dark red the highest.

Medium-low species were also mostly birds, but covered a wider range of families (*Sulidae, Larridae, Fregatidae, Procellariidae*) and included two mammals (*Delphinidae*), the Fraser’s dolphin (*l. Hosei*) and Spinner dolphin (*s. Longirostris*). Two birds—Bulwer’s Petrel (*b. Bulwerii*) and Newell’s Shearwater (*p. Newelli*)—and ten mammals were ranked medium vulnerability. The mammals were mostly *Delphinidae* (six species), but there were also *Kogiidae*, *Balaenopteridae*, and *Ziphiidae* species. Seven mammals, five birds, and one turtle, Olive Ridley (*l. Olivacea)* had medium-high vulnerability. The birds in this category were from four families (*Hydrobatidae*, *Rallidae*, *Anatidae*, and *Diomedeidae)*, as were the mammals (*Delphinidae, Kogiidae, Ziphiidae*, and *Physeteridae*). Finally, all three taxa were represented in the high vulnerability group, including four out of five turtles, two birds—Short-tailed Albatross (*p. Alabatrus*) and Hawaiian Petrel (*p. Sandwichensis*)—and six mammals—Hawaiian Monk Seal (*n. Schauinslandi*), North Pacific Right Whale (*e. Japonica*), and four *Balaenopteridae*.

Confidence in vulnerability groups.

Figure 6 shows confidence in final species scores and vulnerability categories based on uncertainty in the trait data collected. Confidence intervals are shown for each taxon—birds, mammals, and turtles—in 1a, 1b, and 1c, respectively. For 22 of the 33 bird species, the species confidence interval fell within the assigned vulnerability group, showing that despite uncertainty in some species trait data (i.e., ranges provided for possible mass, population, or longevity of a species), there can be confidence in the final vulnerability group. The eleven species that had confidence intervals spanning multiple vulnerability groups had means close to the category bounds and comparatively large standard errors. No birds spanned three vulnerability categories.

Of the 25 mammals, roughly half (12) had confidence intervals that spanned multiple vulnerability groups, and one mammal spanned three categories. Importantly, confidence in the rankings for the five most vulnerable mammal species were high. There was more uncertainty for mammals in the medium and medium-high categories. Unlike with birds this was more driven by high uncertainty in trait data than by mean vulnerability scores lying close to the category cut-offs. Additionally, the score ranges for medium and medium high species were smaller. Nonetheless, it is important to note that for some of these species, confidence in the given vulnerability ranking is still high as the standard error bar barely included a second vulnerability group.

Confidence in the turtle species vulnerability groups is high, with all four high vulnerability species' standard errors contained within that category. Only the Olive Ridley's possible vulnerability crosses two categories: medium-high and high.

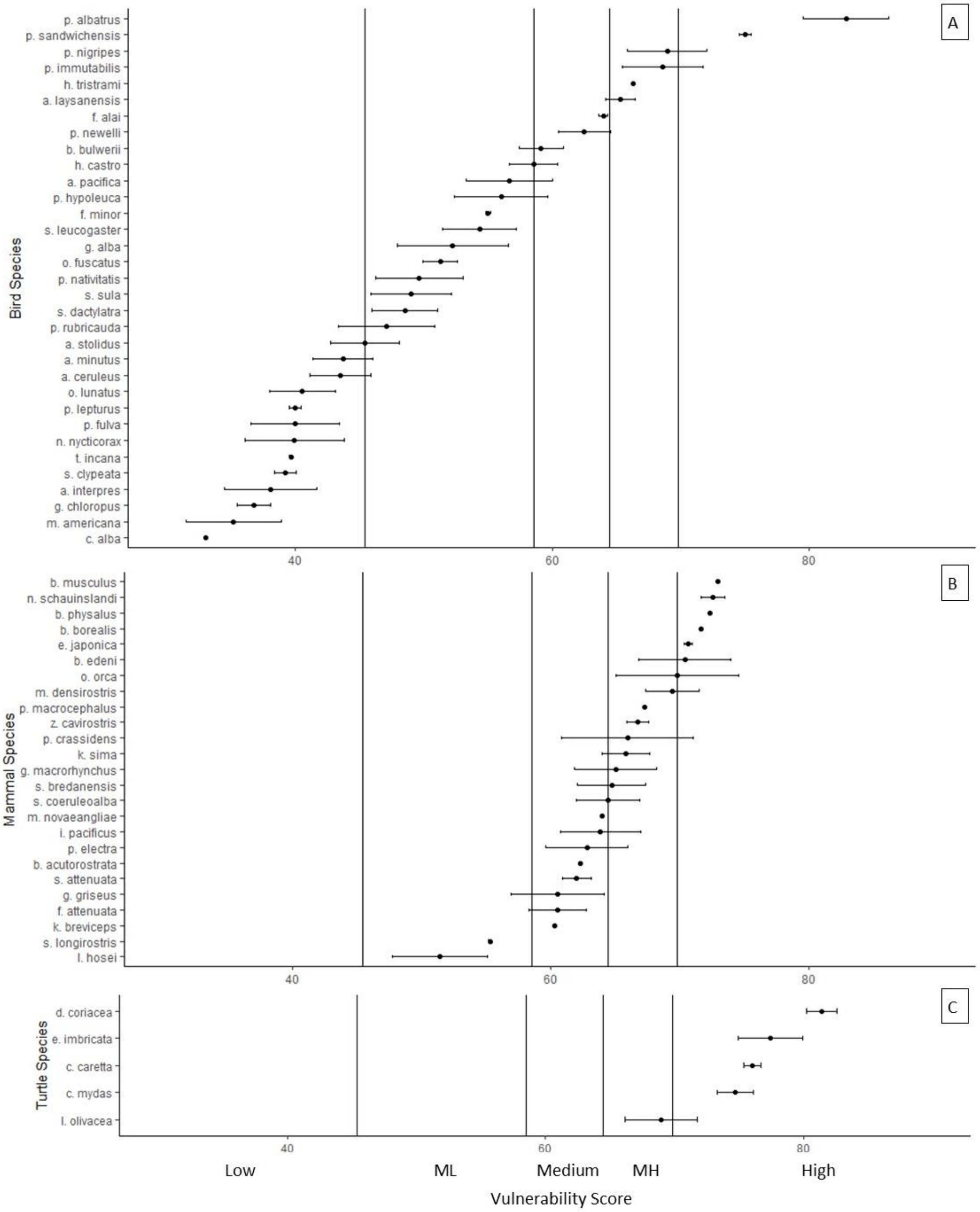

Figure 6. Confidence Intervals for Each Species Vulnerability Score. (A) confidence intervals for bird scores, (B) scores for mammals, (C) scores for turtles.

Correlation between traits. Figure 7 shows the magnitude and direction of correlations between traits. P-values are given for indicators with statistically significant correlations. (p<0.5). Habitat number was most correlated with other traits, having significant, but weak negative correlation with IUCN Red List category, longevity, distribution, generation length, and mass, and a significant, weak, and positive correlation with population (note correlation with habitat specialization is the inverse direction of correlation with habitat number). Generation length had a significant, but weak, negative correlation with population, and had stronger, positive correlations with IUCN status, distribution, and longevity. Population had significant, negative correlations with egestion, IUCN status, and longevity. Mass had a significant, and strong, positive correlation with longevity, and a significant, strong correlation with prey preferences. Finally, IUCN status had a significant, positive, correlation with longevity.

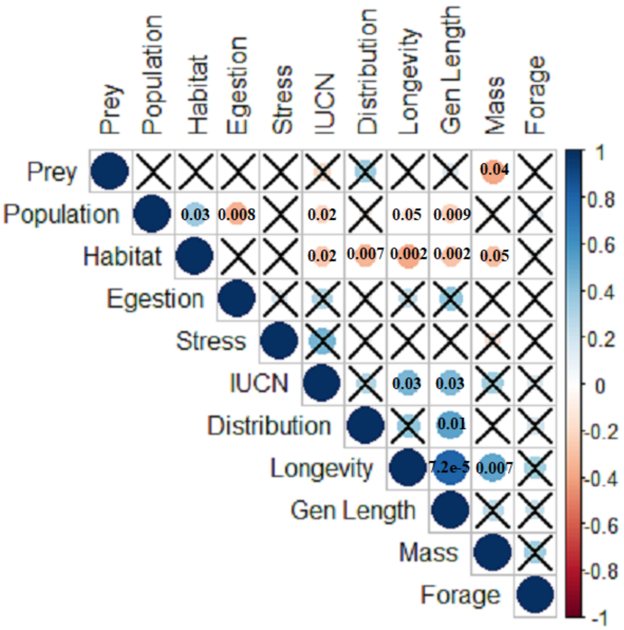

Figure 7. Correlation Coefficients Between Traits. Blue indicates positive correlations. Red indicates negative correlations. Larger circles indicator greater magnitudes of correlations. P-values are provided where correlation is significant and an "X" over the circle indicates the correlation is not statistically significant.

To test for redundancy, vulnerability scores were recalculated removing correlated traits in the same dimension. Therefore, I removed prey and mass from the sensitivity traits and each of the four population resilience traits. Removal of each trait changed the ranking of eight (removing mass or prey from sensitivity score) to 28 species (removing population from resilience score), which confirmed their lack of redundancy, even as some traits showed some correlation. As a result, all 11 original traits were kept in the development of the final ranking (See supplementary materials for more detail).

Discussion

Results emerging from my analyses align with existing literature. The four sea turtle species that were categorized as having high vulnerability, Green Sea Turtles (*c. mydas*), Hawksbills (*e. imbricata*), Loggerheads (*c. caretta*), and Leatherbacks (*d. Coriacea)* are all known to be highly impacted by both ingestion and entanglement throughout their life cycle (Tagliolatto et al. 2020; Gündoğdu et al. 2019; Thiel et al. 2018; Aguilera et al. 2018; Triessnig et al. 2012). Coupled with their slow reproductive turnover and long-life expectancy, their high vulnerability is expected (IUCN, 2022). Further, the Olive Ridley was expected to have lower vulnerability than the other turtles, because of their pursuit of mobile prey that are less frequently mistaken for plastics, reducing their ingestion rates (Bjorndal et al. 1994; Abreo et al. 2019; Lazar and Gracan, 2011). However, they are still long-lived and suffer from entanglement during fishery interactions (Yaghmour et al. 2020).

Broadly, mammals were less vulnerable than sea turtles and more vulnerable than sea birds. Oceanic dolphins generally had the lowest vulnerability, followed by beaked whales, and then baleen whales, with the one pinniped species holding the highest vulnerability group. This broad pattern in the ranking reflects findings in the literature (Thiel et al. 2018, Im et al. 2020, Alexiadou et al. 2019, Puig-Lozano et al. 2018). There is extensive evidence in the literature of the Hawaiʻian Monk Seal's high vulnerability to macroplastic pollution, through ingestion and entanglement (Henderson 2001, Donohue and Foley 2007). This species has a small population and, like other pinnipeds, is vulnerable to fisheries-based plastics due to their prey preferences, foraging behaviors, and curiosity (Hofmeyr et al. 2006). Similarly, research indicating the Atlantic Right Whale is highly vulnerable to entanglement, supports the categorization of the Pacific Right Whale as highly vulnerable (Moore and Van der Hoop 2012). There are a few unexpected results for mammals. For example, the categorization of pygmy sperm whales as medium, and dwarf sperm whale as medium-high is unexpected, as these two species are very similar and unique (McAlpine, 2018). The primary difference between these species scores were driven by differences in listed IUCN threats, and the standard errors around these species scores were high, giving lower confidence in their final categories. For poorly studied species, such as the pygmy and dwarf sperm whales, even though their rankings are more moderate, more research is needed to better understand their potential risk.

Overall, birds were identified as the least vulnerable taxa to macroplastic pollution, but they also had the largest range in vulnerability. Generally, ducks and shorebirds had the lowest vulnerability; followed by noddies, terns and boobies; and shearwaters, petrels and albatross were the most vulnerable groups. This result can be explained by the relatively low documented rates of ingestion in ducks, noddies, and terns, while ingestion rates in albatrosses, petrels, and shearwaters are high (Fry et al. 1987; Sileo et al. 1990; Rapp et al. 2017).

A recent study showed that shorebird species ingest plastics as well and may just be understudied (Flemming et al. 2022). Though these species received low vulnerability scores, they also ranked high for species' sensitivity. This result suggests their life history may make these species more likely to ingest plastics than the previous attention in the literature suggests, supporting the findings of Flemming et al. (2022). My results suggest that lower exposure rates and higher population resilience may reduce vulnerability at the population or species level.

The categorization of the petrels and albatrosses in the medium-high and high vulnerability groups also makes sense given high ingestion rates for nocturnal petrels and albatrosses are well documented (Sileo et al. 1990; Rapp et al. 2017). This result also aligns with the results from a trait-based assessment conducted for seabirds in California (Good et al. 2020). Though research on the short-tailed albatrosses (*p. albatrus*), the most vulnerable bird, is limited, Donnelly-Greenan et al. (2018) found high rates of ingestion in chicks and adults were likely to cause damage to the gastrointestinal tract. Their assignment to the high vulnerability group also makes sense in light of their long-life expectancy and small population (IUCN, 2022).

For a few species, the assigned vulnerability group did not align with what would be expected from the literature. For instance, wedge-tailed shearwaters (*a. Pacifica*) and Newell's Shearwater (*p. Newelli*) have high documented rates of ingestion but were characterized as having medium-low and medium vulnerability, respectively (Kain et al. 2016). However, both still received high scores for species sensitivity. It is possible that their population resilience reduces their overall vulnerability. Alternatively, the medium-high categorization of the Laysan Duck was greater than expected, as there is little evidence in the literature of species' sensitivity (ingestion or entanglement of individuals). However, their extremely low population resilience is increasing their vulnerability rank. Plastic interactions may also be occurring and simply poorly documented in the literature, as seen with other shorebirds (Flemming et al. 2022). Research to explore these discrepancies is warranted.

Confidence in vulnerability categories. To my knowledge, this is the first effort to integrate uncertainty into a relative trait-based vulnerability index. Typically, studies provide a mid-range score (e.g., 3/5) for unknown data, or provide a best estimated score (Woodyard et al. 2022; Chin et al. 2010; Foden et al. 2013). My results indicate that the ranking system can handle some uncertainty and provide precise vulnerability rankings for the species of interest. However, at an individual species level, there was less confidence in species' vulnerability rankings if there was a lot of uncertainty in their trait data. This had a bigger impact on vulnerability rankings for species in the low-medium to medium-high categories than on those in the low or high categories. For example, the pygmy and dwarf sperm whales are closely related, understudied species that researchers may expect to have similar vulnerability. They received different vulnerability rankings, but both have large error bars, and their possible score ranges overlap substantially. Therefore, this indicates that for species with a lot of trait data missing or with broad estimates, specific vulnerability estimates may be inaccurate.

Application of the Hawaii Vulnerability Index. The vulnerability index and ratings presented here can be used to prioritize species and geographic areas for improved management, monitoring, and plastic mitigation efforts in the Hawaii EEZ. Clean-up efforts can focus on marine regions, beaches, and nesting areas more frequently used by these species. Additionally, depending on the species, different upstream management efforts may provide more benefit. For instance, the most vulnerable mammals are disproportionately impacted by fisheries-based plastics (Puig-Lozano et al. 2018, Thiel et al. 2018, Boland and Donohue 2003). Therefore, focusing on fisheries interactions—derelict gear removal, fishing for debris programs, or regulations on gear types—may provide greater outcomes for these species. Alternatively, addressing plastics, such as plastic bags, that are more often mistaken for food may provide greater benefits for species that feed on squid and other prey that resemble soft plastics (Poli et al. 2015). Finally, long-term monitoring and population studies should focus on the species identified in the high and medium-high categories. These species will also provide a good indicator for overall ecosystem impacts of plastic pollution.

Beyond the value of the final vulnerability score, there is additional value in considering the quintile score of species for each dimension of vulnerability. For instance, if a species has low likelihood of exposure but is highly sensitive with a vulnerable population, then it may be important to monitor for changes in local plastic exposures. Range wide exposure rates may not reflect the exposure of local populations, and these could be due to changes in local plastic use, marine activity pressures, or changing currents. Alternatively, looking at species sensitivity may highlight cases where species that are less vulnerable are getting more attention due to high individual interaction rates, but a species with lower observed interactions may be more vulnerable because of a less resilient population.

Broader implications. In the theory and practice of conservation, species recovery efforts are typically focused on single species versus threat mitigation across species. Given the ubiquitous nature of plastic pollution in our oceans, focusing on the threat itself (plastic pollution) using TBAs offers a promising approach for designing effective mitigation strategies. My work represents an important first step in this direction. In particular, the development of this vulnerability index exemplifies the potential of trait-based approaches for macroplastic pollution research across the globe. Though this index focuses on Hawaiʻi, many of the species evaluated have social and ecological importance across the globe. Further, the methods applied here could be expanded for global analyses, or to include more taxonomic groups. Such indices could be used to identify understudied species, explore community and ecosystem level effects, and choose the species best suited for long-term monitoring.

These indices could also be used to inform species management and plastic mitigation efforts. At the local scale, managers and decisionmakers could apply this framework similarly to understand the impact of plastics on local ecosystem health and direct local priorities. At a global scale, this framework could be implemented to identify species requiring international cooperation. In both cases, this could inform policy priorities for ecological outcomes, such as regions and plastics to target for mitigation. Finally, further research into TBA approaches for macroplastic pollution could lead to their implementation for more advanced ecological risk assessments and the development of regulatory thresholds.

Limitations. There are important limitations to TBAs that must be acknowledged with their implementation. First is data availability. One reason little is known about the consequences of macroplastic pollution on marine biodiversity at higher levels of biological organization is that these species are difficult to study at both the individual and population level (Bucci et al. 2020; Murphy et al. *in review*). As a result, trait data are often limited in accuracy and precision. The three scenarios for standard error exemplify that confidence in vulnerability indices are closely tied to confidence in trait data.

Even when trait data are available, there is still uncertainty associated with the scoring of some traits. For example, plastic density within a species' range is the most important indicator of exposure. However, species range maps are coarse and assume individuals are evenly distributed throughout their range. Additionally, global plastic distribution maps are limited to surficial densities (Eriksen et al. 2014). Higher resolution data on plastic and species distribution could improve these predictions, yet TBAs will always be limited by the quality of their assumptions, and these should sufficiently be considered with the literature. However, when high-quality trait data are lacking, coarser vulnerability groups can be developed to build greater confidence in vulnerability ratings (i.e., split species into three categories: low, medium, and high, instead of five). These provide less resolution, but still identify the most vulnerable species.

There are also limitations specific to multi-taxonomic indices. There is causal data in the literature about how different feeding and foraging behaviors among birds affect ingestion rates, but it is more difficult to compare sensitivity associated with feeding and foraging behaviors across taxa. This creates risks of inaccurate clumping of taxa and these challenges increase when trying to compare more distantly related taxa.

Future research. To improve the value of TBAs for marine macroplastic pollution, four areas of future research are needed. First, more research on marine biodiversity life history is needed to improve the quality of trait data. Higher accuracy of TBAs for macroplastic pollution would expand the opportunities for applying TBAs to other stressors faced by marine biodiversity. Second, additional research is also needed to understand the relationship between traits and plastic pollution as well as on TBA methodology broadly to improve methods for weighting traits and scoring species. The current approach weights all traits equally within a vulnerability dimension (i.e., likelihood of exposure, species' sensitivity, or population resilience); however, it is clear some traits have more influence on vulnerability than others and results would be more accurate if they were not all weighted equally. Third, strategic, placed-based population, species, and community level research on the physical impacts of macroplastic pollution are needed to validate vulnerability indices on the ground. Validating indices would allow for these approaches to be applied more broadly with greater confidence in the traits included and accuracy of outcomes. Finally, more research is needed to integrate TBAs into ecological and other risk assessments, to increase their value for policy development and decision making.

Conclusion

Marine macroplastic pollution has significant physical consequences on marine biodiversity. Given that little is known about the impacts of macroplastic pollution at population, species, or community levels, trait-based approaches provide a salient method for inference across biological organization. Here, I present the first multi-taxonomic index for vulnerability to macroplastic pollution ingestion and entanglement. This work provides insight into the most vulnerable marine megafauna—birds, turtles, and mammals—in Hawai‘i, showcases the value of the framework put forth in Chapter four and exemplifies the potential for TBAs in research and managing marine plastic pollution more broadly.

CHAPTER 6

CONCLUSIONS

Marine plastic pollution is an urgent environmental challenge with far-reaching ecological and socio-economic consequences. Despite extensive management efforts, marine plastic pollution continues to increase along with its negative consequences. In this dissertation, I explore applied research questions to better understand how marine plastic pollution can be addressed in a cost-effective, equitable manner that achieves the desired socio-economic and ecological outcomes. In my first chapter, I introduce the challenges plastic pollution poses for human and natural communities. In my second chapter, I present the idea of the *plastic-scape,* a novel model for understanding plastic pollution using a seascape ecology approach. In my subsequent chapters, I develop and test frameworks to improve marine plastic pollution management. In chapter three, I present a framework to estimate the total cost of marine plastic pollution interventions and use case studies to highlight its functionality and key factors that influence cost. In chapter four, I present a framework for developing indices of biodiversity vulnerability to macroplastic pollution using a trait-based approach. Finally, in chapter five, I use the multi-taxonomic vulnerability framework I present in chapter four to develop the first multi-taxonomic vulnerability index for ingestion and entanglement of macroplastic pollution – assessing birds, marine mammals, and turtles in Hawaii. Combined my dissertation research contributes to the larger body of knowledge on the social-ecological impacts of MPP and MPP management, presents new tools for understanding and mitigating these socio- ecological effects, and can ultimately contribute to more cost-effective, and equitable management of MPP and its social-ecological consequences.

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APPENDIX A

SUPPORTING INFORMATION FOR CHAPTER THREE

Table A1.

Papers Returned for each Web of Science Search

Citation	**Summary of key findings**
Economic AND Marine AND Litter (Total returns: 71)	
Beaumont et al., 2019	Review of the global ecological, social, and economic impacts of marine litter. Estimates an economic cost of $3,300–$33,000/tonne of marine plastic annually.
Agamuthu et al., 2019	Literature review of the impacts of plastic pollution and global initiatives to address plastic pollution.
Cordier and Uehara, 2019	Removing 15% of plastic debris from the ocean every year from 2020-2030 (135 million metric tons of plastic total) would cost (sic)492 billion-(sic)708 billion, which represents 0.7%-1.0% of the world GDP in 2017 (based on Ocean cleanup).
Krelling et al., 2017	Litter may potentially reduce local tourism income by 40% - 8.5M/year at 2 Brazilian subtropical beaches.
Schuhmann et al., 2016	Willingness to pay differences between clean and littered beaches showed stronger aversion to beach litter in Barbados.
Wagner et al., 2016	Resulting litter from curbside collection – clean-up cost is 0.17-0.79$ per piece of litter, loss in recycling revenue is $3.92-19.25/household/year.
Driedger et al., 2015	Identifies impacts of marine plastics to industry (fisheries and tourism) and human wellbeing (for coastal communities).
Oosterhuis et al., 2014	Literature review of the cost-effectiveness of economic instruments for marine litter control. Shows context of implementation matters.
Ostberg et al., 2012	Reviews willingness to pay for reduced littering but grouped with noisiness of beached in evaluation.
Ryan et al., 2009	Large spatial and temporal heterogeneity in the amounts of plastic debris, with greater loads close to urban areas. Clean-ups of low-density debris are costly.
Nash, 1992	Impacts of debris on subsistence fisherman. >50% of gill net fishing has debris.
Cost AND Ocean AND Plastic (Total returns: 59)	
Choi and Lee, 2018	Willingness to pay for removing microplastics. Seoul: $9.8 M or 2.59/year/person.
Hermawan et al., 2017	Estimates the impact of marine plastic pollution on Selayar fisherman. Vessels repairs cost 192.9M rupiahs/year and gear repairs cost 156.2M/year.
Raubenheimer and McIlgorm, 2017	Costs of plastic pollution: removal by coastal communities, repairs to equipment/infrastructure, reduced harvests, reduced tourism. Montreal agreement raises cost of implementation and cost-benefit. Cost should include ecosystems and

	environmental values only including these allows for consideration of no-action vs. action.
Cost AND Marine AND Plastic (Total returns: 176)	
Abbott et al., 2019	Review economic literature to assess potential of different economic policies for targeting marine plastic pollution, including bans and standards, extended producer responsibility, price-based policies, and interventions grounded in behavioral economics and psychology. Consider feasibility in particularly in coastal developing nations and how policies link with informal sector.
Qiang et al., 2019	Tourism revenue can be increased by 32.23% along east China sea if beaches are cleaned up.
Cost AND Marine AND Debris (Total returns: 82)	
Campbell et al., 2019	Over 10 years in New Zealand, 78,370 new beach injury claims were lodged, and 82,891 active beach injury claims were accepted and managed due to beach litter. The average claim was NZ$450.
Shen et al., 2019	More than 74% of interviewees are willing to volunteer for 1.5 days/month to clean to have clean beaches. Value is $1.08-1.4/per person for contingent method and $1-1.07 for choice method.
Brouwer et al., 2017	Examined how beach visitors in Greece, Bulgaria and the Netherlands perceived marine litter. Greek beaches were perceived as most dirty, followed by Bulgarian and Dutch beaches. Bulgarian beach visitors were most willing to participate in clean-ups, followed by Greek visitors, with Dutch visitors being least willing to volunteer. Bulgarian beach visitors were willing to pay most for beach clean-ups.
Leggett et al., 2014	Discusses the direct annual costs spent on cleaning up litter in California cities.
McIlgorm et al., 2011	Examine costs associated with marine debris. Present a marine debris cycle model to discuss the costs and benefits of prevention, clean-up and biodegradable materials. Estimate marine debris-related damage costs $1.26bn/year in 2008 terms to marine industries in the 21 economies of the Asia-pacific rim.
Ballance et al., 2000	Visitors to the Cape Peninsula in South Africa would pay more than 7x the average trip cost for a clean beach.
Economic AND Marine AND Debris (Total returns: 132)	
Williams et al., 2016	Examined tourism preference for Colombian North Caribbean beaches. Marine litter placed most beaches into the poor scenic category and clean-up efforts would improve their ratings.

Jang et al., 2014	Estimate an economic loss for South Korea tourism of $29-37M due to plastic pollution.
Economic AND Ocean AND Debris (Total returns: 59)	
Schuyler et al., 2018	States with consumer deposit legislation had 40% lower container pollution on beaches than states without legislation. Greater impacts were seen in areas with lower socio-economic conditions and higher waste loads in the US and AUS.
Cost AND Marine AND Litter (Total returns: 62)	
Wyles et al., 2016	Litter can undermine the psychological benefits of the coastal environment.
Economic AND Marine AND Plastic (Total returns: 126)	
Economic AND Ocean AND Plastic (Total returns: 60)	
Marine AND Plastic AND Intervention (Total returns: 27)	
Economic AND Ocean AND Litter (Total returns: 28)	
Cost AND Ocean AND Debris (Total returns: 38)	
Cost AND Ocean AND Litter (Total returns: 22)	
Ocean AND Plastic AND Intervention (Total returns: 10)	
Marine AND Debris AND Intervention (Total returns: 33)	
Ocean AND Debris AND Intervention (Total returns: 12)	
Marine AND Litter AND Intervention (Total returns: 23)	
Ocean AND Litter AND Intervention (Total returns: 7)	

Table A2

Detailed Summaries of the Citations that Informed the Comparative Case Studies

Cost summary for intervention	**Location**	**Citation**
Coastal clean-ups in developed cities over one-year		
Direct costs: Annual costs for beach clean-up in CA. San Diego: $342,165, based on two 3-hour volunteer clean-ups. Opportunity cost/volunteer = $42.72 (likely underestimate) Long Beach: $1,837,390. $892,223 in labor, $845,175 in equipment, and $100,000 for renourishment.	US west coast cities	Leggett et al., 2014
Avoided costs: 50% decrease in marine debris at 31 beaches in Orange County could generate $67M in benefits over 3 months. 75% reduction in debris from six beaches near outflow of LA River would increase visitation by 43%, for $53 million in benefits.	US west coast cities	Stickel et al., 2012

Direct costs: Average: €139,043.21/municipality (66% workforce, 5% administration, 10% collection, 7% materials, 12% disposal) *Avoided costs*: Coastal tourism is €7 to €11 billion. 90% of municipalities cleaned beaches because they were tourist destinations, 52% for public health reasons, 17% local business *Non-monetary benefits:* 22% cleaned beaches in wildlife reserves *Non-monetary costs:* 89% landfill some waste, 17% incinerate waste	United Kingdom	Mouat et al., 2010
Direct costs: R3.5 million (Cape Metropolitan area for 1994-1995) *Avoided costs:* R3 million, 85% of surveyed tourists would not visit a beach with more than 2 large pieces of plastic.	Cape town, South Africa	Ballance et al., 2000
Coastal clean-ups in developed cities, long-term		
Direct costs: Increased by 37.4% over 10 years, from average of €87,037.005 to €139,043.21/municipality. Increases in disposal costs, amount of litter, labor costs, and maintenance and fuel for vehicles. *Avoided costs*: Reduced tourism value/unit cleaned, because of increased tourist's expectations for "clean".	United Kingdom	Mouat et al., 2010
Direct costs: $30 million spent on coastal cleanups in municipalities after typhoons and floods over 8 years	South Korea	Han et al., 2010
Coastal clean-ups in high-income cities over one year		
Direct costs: €17.9 - €18.8M. 66% workforce, 10%collection, 12% disposal, 7% materials, 5% administration *Avoided costs*: 92% cleaned litter from high use beaches vs. 43% from low-use beaches and 12% from isolated beaches Beaches cleaned: 77% Sandy, 54% shingle, 31% rocky, 10% mudflats, 2% salt marshes. Sandy beaches are easier and cheaper to clean. Rocky beaches are expensive.	United Kingdom	Mouat et al., 2010
*Direct costs: $*9M for beach clean-ups, cleaner-ship in ports/harbors, 19 seabed litter clean-ups, and education in 42 admin units. *Indirect offset:* Job creation *Avoided costs:* Fishing industry, tourism *Non-monetary benefits:* improved wildlife health and aesthetics *Non-monetary costs:* Incinerate most waste *Equity:* Workers were senior citizens and/or low-income.	South Korea	Han et al., 2010
Waste to energy developed municipality, 20 years		

Lancaster,PA: *Direct costs*: high capital investment ($135M); *Offset costs:* tipping fees ($465-517M), sale of metals (128,800 tons), sale of renewable energy ($256M); *NM benefits:* Reduce greenhouse gas emissions, reduced land use **Pinellas County, FL:** *Direct costs:* Capital investment is $58M/year, Processing waste (labor and operating costs) is $28/ton *Offset costs:* $80M/year. Tipping fees ($37.50/ton), electricity sales, capacity payments from the energy company, and recycling revenue **Spokane, WA:** *Direct costs:* Capital investment. *Offset costs:* Recovered >200,000 tons of ferrous metal, produced 2.8 billion KWH of electricity. (paid off investments in 20 years) **Portland, ME:** *Direct costs:* Capital investment ($93M) *Offset costs:* 100,000 megawatts-hours of electricity annually Paid of capital investment in 27 years. *NM benefits:* Environmental benefits (ISO 14001 certified for excellence in environmental management	United States	Crawford, 2008
Direct costs: €68.18/tonne for a 250,000 tonne plant. Capital assets: land, infrastructure (high costs). Consumables: raw materials, Labor: Salaries, training. Overhead: Maintenance of facilities and equipment. *Indirect and non-monetary costs*: €38.73− 48.21. *Indirect costs:* Human health and environmental costs of pollutants (mainly NOx and SO2). *Non-monetary costs:* Unsightliness and odors (Euro8/ton), CO2 emission. *Offset costs*: Gate fees (70-80% of offset), energy sales (20-30% of offset), recycling of metal post-incineration, and sale of combustion residuals. *Avoided costs:* Costs of landfilling (€ 30.75–€38.66/ton) and coal powered energy plants (€48.88 - 84.27/ton). *Non-monetary benefits:* GHG sink: 1 tonne of MSW reduces consumption of oil by 1 barrel and coal by 0.26 tonnes. Reduces methane emissions. Factors influencing cost: Plant efficiency, composition of the waste stream (less moisture, more plastics = more energy). Technological progress is assumed to reduce the total private cost of the facilities by 1.5% annually.	United Kingdom	Jamasb & Nepal, 2010
Non-monetary benefit: GHG sink; due to advanced technology and high quality MSW, with low water content.	European Union	Yang et al. 2012

Indirect costs and non-monetary costs: Low; little air pollution so low health care costs, and impacts to human wellbeing and the environmental are low *Offset costs*: Efficient energy recovery and production *Non-monetary benefits:* GHG sink	European Union	Lombardi et al. 2015
Direct costs: Capital assets, O&M *Indirect costs:* .88Euro/MWH h. Human health costs of air pollution (NOx, SOx, dixoins, heavy metals), water pollution 2.23 Euro/ton (CSERGE, 1993). External costs of energy supply (16.5Euro/ton) *Offset costs*: power sold to grid *Non-monetary costs*: $43.75/ ton due to high opposition. GHG emissions (fuel from waste transport, energy use and process emissions in treatment, avoided emissions), impacts to QOL (noise, aesthetics), land use change	Greece	Mavrotas et al. 2015
Direct costs: 103 Euro/ton. Capital investments, abatement technology. *Offset costs*: 21 Euro/ton, 12% net profit over 25 years (minimum is 648 ktonne). *Indirect/non-monetary costs:* air pollution (47 pollutants) 17.26 E/ton, chemical waste 28.69 E/ton *Non-monetary benefits:* energy saving (22.62E/ton), materials function 5.76E/ton. *Total* = 97/ton. Cheaper technology means lower direct costs buts higher environmental costs	Netherlands	Dijkgraaf & Vollebergh 2003
Pre-treatment to reduce moisture content of waste increased cost/ton	Italy	Consonni et al. 2005
Waste to energy developing municipality, 20 years		
Direct costs: Capital cost (may be too high for developing countries), operation & maintenance (requires training and technical skills) *Offset costs:* Electricity generated (low quality waste reduces profit potential), tipping fees *Equity*: Electricity produced may be too costly for low-income communities (cost of $3.67 per 0-600KWh)	Ghana	Fobil et al. 2005
Direct costs: Capital investment (75-97CNY/tCO2), operations &maintenance (74-80CNY/tCO2), sales tax/surcharges (1.22-1.54CNY/tCO2) *Offset costs:* gate fees (108-175CNY/tCO2), energy production sales (96-99CNY/tCO2), heating benefit (32.83 CNY/tCO2 in northern cities) Costs are positive without gate fee offsets but negative with gate fees.	China	Wang et al. 2016

Direct costs: Capital investments ($110-329M); operation and maintenance: increased by high risk of corrosion, because of MSW content. Costs aren't sustainable for most cities. Organic-rich waste means lower burn temperatures: less energy produced, more smoke produced, worse air quality, lost renewable resources (organic substances) *Indirect costs*: Air pollutants (primarily dioxins) which have been linked with health problems (16% of plants don't meet national standards, 78% don't meet EU standards though they claim to); open dumping of fly ash which leaches contaminates associated with water pollution, GHG source *Potentials*: Could shift from GHG source to sink, and have lower indirect costs with improved waste segregation and newer technology.	China	Zhang et al. 2015
Direct costs: Capital investments are $73.35M (processes 1000 tons/day). Includes: equipment, risk management, construction, land use, preparation funds, loan interest. Operation costs are $7.14M/year. Includes: raw materials, plant power consumption and auxiliary fuel (31% of cost), labor (15%), depreciation & maintenance (37%), environmental expenses (fly ash handling, environmental monitoring, bottom ash processing, leachate treatment). *Non-monetary benefit:* GHG could be reduced when landfill gasses are considered and if incineration heat is used for energy. A*voided cost:* Kills pathogens, landfilled waste reduced by 90%, O*ffset costs:* electricity sales and waste disposal subsidies ($13.04/ton) Total is $15.59M/year; break even at 13 years	China	Xin-gang et al. 2016
High-water content waste burns at low temperature affecting costs below: *Indirect and non-monetary costs:* more air pollution and particulates *Direct costs:* Excess auxiliary fuels needed for complete combustion *Offset costs:* Less energy is produced. *Non-monetary cost:* GHG source in all investigated Chinese cities, except Urumqi. Plastics were the biggest contributor to GHG emissions.	China	Yang et al. 2012
Energy content of MSW in China is low (4–7 GJ/Mg). Therefore:	China	Lombardi et al. 2015

Direct costs: more auxiliary fuel needed to burn waste
Offset costs: lower energy production.
Additional considerations: Smaller plants are less efficient than large plants, but large plants require larger investments

Table A3

A blank Version of the Framework Presented in this Manuscript

Intervention:
Objective:
Scale (spatial and temporal):

Stakeholders	Actions and direct costs		Indirect and non-monetary costs	Recovered costs, monetary benefits & NM benefits
#1	Purchase capital asset 1	$XXXXX	Indirect costs:	Recovered costs:
			NM costs:	Monetary benefits:
				NM benefits:
#2			Indirect costs:	Recovered costs:
			NM costs:	Monetary benefits:
				NM benefits:
#3			Indirect costs:	Recovered costs:
			NM costs:	Monetary benefits:
				NM benefits:
#4			Indirect costs:	Recovered costs:

	NM costs:	Monetary benefits:
		NM benefits:

Net costs (calculate using the cost equation):
Stakeholder 1:
Stakeholder 2:
Stakeholder 3:
Stakeholder 4:

Equity:

APPENDIX B

METHODS AND RESULTS FOR VULNERABILITY INDEX DEVELOPMENT

Methods for calculating species trait values and scores

This section provides detailed descriptions of how a species score was calculated for each trait.

Distribution (average plastic density in range). Distribution scores were calculated using ArcGIS geoprocessing and were measured as the average number of macroplastics per km^2 within the species range. Plastic occurrence data was taken from Eriksen et al. (2014). Data for all class sizes greater than 5mm were aggregated to create a macroplastic density map. Macroplastic density maps were uploaded as shape files with polygons representing different plastic density ranges. Species were downloaded from the IUCN red list website. Species range maps and plastic occurrence maps were loaded into ArcGIS. The maps were cropped so only the intersect areas were shown. Then the proportion of the species range in each plastic density group was calculated to determine the average density of plastic in the species range per km^2. For bird populations, the area of the intersect map was divided by the total range area to calculate the terrestrial range, and the plastic density in the terrestrial portion of the range was assumed to be zero. All density scores were assumed to have a standard deviation of 2.5% to account for some errors in species range and plastic density maps.

Plastic density scores were then calculated as a quintile, with a score of one representing species with lowest plastic density in their range and a score of five represented species with the high plastic density. Quintiles and subsequent scores were calculated in RStudio (code provided below). To account for standard error in estimates, I produced 1000 estimates for plastic density /km^2, assuming a normal distribution around the mean and with the standard deviation provided. The 6300 distribution values produced (1000 runs * 63 species) were then used to calculate the quintile cut-offs for the possible scores of one to five for each species. Then 1000 scores were calculated for each species, by scoring the 1000 random draws against these quintile cutoffs. This produced a mean score of one to five for each species with a standard error for this score.

Longevity (life span). I sourced longevity data first from IUCN when available. If it was not available from IUCN, I developed a longevity range for each species based on data available from the Animal Diversity Web, for all taxa, and NOAA for turtles and mammals or Birds of the World for birds. If a single longevity score was provided, I assumed a standard deviation of 2.5%. If a single longevity score was provided but low certainty was expressed, I assumed a standard deviation of 10%. For species with a range in longevity, I assumed the range had a 95% percent confidence interval with a normal distribution, making the middle of the range and the minimum and maximum values each being two standard deviations from the mean. Finally, if a record high was provided with no other longevity data, I assumed a standard deviation of 10%, with the longest live record being two standard deviations above the mean.

Longevity was then scored using quintiles, giving each species a score of one to five where a score of one represented the shortest-lived species. Quintiles were calculated in RStudio, using the same methods provided above for distribution.

Body morphology (mass). I sourced mass data first from IUCN when available. If it was not available from IUCN, I developed a mass range for each species based on data available from the Animal Diversity Web (for all taxa), and NOAA (for turtles and mammals) or Birds of the World (for birds). I then calculated mean and standard deviations for each species' mass using the same method described for population. For the Hawaiian Coot, I used American Coot mass estimates. For other species where mass estimates were not available, I used the average mass for its taxa and included a standard deviation of 10%. I then calculated quintile scores in R, where the smallest species received a score of five, using the same methods as described for distribution.

Foraging and feeding behavior. Data on feeding and foraging behavior were taken from IUCN first, followed by the Birds of the World database for birds and from NOAA and Animal Diversity Web for mammals and turtles. Feeding and foraging behavior was then ranked from one to five based on the qualitative categories shown in Table B1. Assumptions on relative sensitivity based on feeding and foraging behaviors were drawn from peer-reviewed literature. For species where multiple feeding strategies were listed an average score was calculated based on the score associated with each feeding type used, and averages were rounded up to the nearest integer. No uncertainty was included in score estimates.

Table B1

Scoring for Feeding and Foraging Behaviors

Score	Qualitative feeding and foraging behavior data
1	Pick and probe; under water pursuit; pursuit diving; stealing in flight
2	Plunge diving; biting
3	Dabbling; swallowing prey whole; deep diver
4	Flutter on surface; dipping; grazing
5	Surface seizing; scavenging; filter feeding

Prey preferences. Data on prey preferences were taken from IUCN first, followed by the Birds of the World database for birds and from peer-reviewed literature for mammals and turtles. Prey preference was then ranked from one to five based on the qualitative categories shown in Table B2. Assumptions on relative sensitivity based on prey preferences were drawn from peer-reviewed literature. No uncertainty was included in score estimates.

Table B2

Scoring for Prey Preference

Score	Qualitative data on prey preferences
1	Specialize on food that does not resemble plastics (based on observation in peer-reviewed literature) and does not increase likelihood of interaction with fisheries
2	Generalist on food that does not resemble plastics and does not increase likelihood of interaction with fisheries
3	Generalist that consumes both foods that do and don’t resemble plastics
4	Feeds on species that are driven to the surface by fisheries-species; specializes on species that have some documented resemblance to plastic (cephalopods); feeds on flying fish eggs, which are documented to be laid on plastics
5	Specializes on food that are well-documented to resemble plastics (jellyfish) or that are provisioned by commercial fisheries (e.g., tuna); Consumes human refuse

Egestion potential (regurgitation). Egestion potential was estimated using potential for regurgitation. Ingestion of gastroliths was also incorporated into categorical scoring because species that ingest gastroliths are more capable of breaking up plastics. Data on regurgitation capacity was taken from the Birds of the World databased for birds and from peer-reviewed literature for mammals and turtles. Species egestion potential was then ranked from one to five based on the qualitative categories shown in Table B3. No uncertainty was included in score estimates.

Table B3

Scoring for Regurgitation Potential

Score	Qualitative information on regurgitation capability
1	Regurgitates to young and frequently regurgitates pellets of hard to digest material
2	Regurgitate to young and no information on pellet production, but may produce based on other species in family/genus; regurgitate to young and occasional pellet casting in certain life stages
3	Capable of regurgitation; regurgitate to young but do not produce pellets or ingest gastroliths; ingest gastroliths
4	No information on regurgitation, but may regurgitate to young based on knowledge of other species in the genus/family
5	Anatomical structure makes regurgitation difficult or impossible; don't regurgitate to young or produce pellets; no information on regurgitation to young or pellet production

Vulnerability to other stressors (IUCN threats). Data on species vulnerability to other stressors were collected from the IUCN threats information. IUCN gives each threat an impact score of 1-8, which is based on the scope (i.e., part of the population to the whole population) and severity of impact. For impacts where an impact score was listed as unknown, the highest impact score possible for the given scope was used (i.e., for "minority of population" impacted a score of 4 was used (highest "low impact" score, for "majority of population" impacted a score of 7 was used, and for "whole population impacted" a score of 8 was used. Next, a total stressor score was then produced using the following equation:

Stressor score = Threat_1 impact score + Threat_2 impact score …+ Threat_i impact score (3)

This produced an impact score range of 6 to 41. For each species were data was available a standard deviation of 2.5% around this score was used to account for uncertainty in estimates. For species where no threats data was available, the mean stress score for all species was used, with a standard deviation of 10%. Quintiles for stressor scores were then calculated in R, where the lowest stressor values received a score of one, following the same method as described for distribution.

Abundance (population). Population data was collected from a variety of sources and represents the number of mature adults for all three taxa. When available, data was sourced from IUCN. If not available, NOAA Fisheries data or the Turtle Conservancy were used for turtle population estimates, NOAA Fisheries data or Animal Diversity Web were used for mammal populations, and Birdlife, Birds of the World or Animal Diversity Web were used for bird population estimates. Turtle population data was available as the number of mature females. Therefore, population size was estimated using both the range for the number of mature females and the estimated sex ratio. When a population range was included, it was assumed to be a normal distribution with a 95% confidence interval. Mean was assumed to be the middle of the range. The minimum and maximum of the range were assumed to be two standard deviations from the mean. When a single number was provided, I assumed the standard was 2.5% of the population. I ran 1000 runs for each species, assuming a normal distribution around the mean population value. The 5800 population values produced (1000 runs * 58 species) were used to calculate quintiles that produced possible scores of one to five for each species were the smallest populations received a score of five and the largest a score of one. Five species had unknown populations. Three were listed as unknown and received a score of 3, one was described as “rare” and received a score of 5, the fifth was described as “fairly common” and received a score of 2.

Reproductive turnover rate (generation length). When available, generation length (age at sexual maturity + 1/2 of years of reproductivity) was taken from IUCN. If it was not available from IUCN, I developed a range for generational length for each species based on data available from the Animal Diversity Web, for all taxa, and NOAA for turtles and mammals or Birds of the World for birds. For a few bird species where generation length was not available in Birds of the world, I used peer-reviewed literature to calculate a generation length based on the age of sexual maturity and number of years reproducing. Standard deviation was calculated using the same methods as described for population. Quintiles were then created for generation length in Rstudio, using the same methods as described for distribution, where the shortest generation lengths received a score of 1.

Specialization (habitat use). Habitat specialization was estimated based on the list habitats used for each species in the IUCN data base. This number of habitats was assumed to be a mean number, with a 2.5% standard deviation added to account for error. For species where habitat data was unavailable, the mean habitat number was used with a standard deviation of 10%. Quintiles were then calculated in R using the same methods as described for distribution, with the lowest number of habitats (highest specialization) receiving a score of five.

Risk of extinction (IUCN status). Species were given a score based on their IUCN status, where least concern species were given a score of one, near threatened species a score of two, vulnerable species a score of three, endangered species a score of four, and critically endangered species a score of five. Standard deviation on the scores were assumed to be zero. Data deficient species were given a score of three with a standard deviation of one, meaning that when standard error is considered species could have any IUCN status.

Trait data uncertainty. To evaluate the influence of certainty in trait data on final rankings, I recalculated species scores and estimated uncertainty, first assuming there was a 1.25% standard deviation and then a 2.5% standard deviation around all “known” quantitative trait data (e.g., known populations were coded to have some uncertainty, but IUCN red list status or prey type were not). Under this 2.5% standard error scenario (SD of 1.25%), 14/33 bird species’ standard errors were within their assigned vulnerability group and 19/33 crossed multipled categories (Figure B1A). For mammals in the 2.5% SE scenario, only 6/25 species standard error bars are within the assigned vulnerability group (Figure B1B). For turtles in the 2.5% SE scenario, the four most vulnerable species are still all within the high vulnerability group; however, the confidence interval for Olive Ridley’s spans three vulnerability groups (Figure B1C).

Under the 5% SE scenario (all “known” values were assumed to hav an SD of 2.5%), 14/33 bird species’ standard errors were contained within their vulnerability group and 19/33 crossed multipled (Figure B2A). For mammals in the 5% SE scenario, only the three most vulnerable mammals have their standard errors captured within a single vulnerability group (Figure B2B). For turtles in the 5% SE scenario, the four most vulnerable species are still all within the high vulnerability group; however, the confidence interval for Olive Ridley’s spans three vulnerability groups (Figure B2C). For turtles and birds, these were the same outcome as for the 2.5% SE scenario.

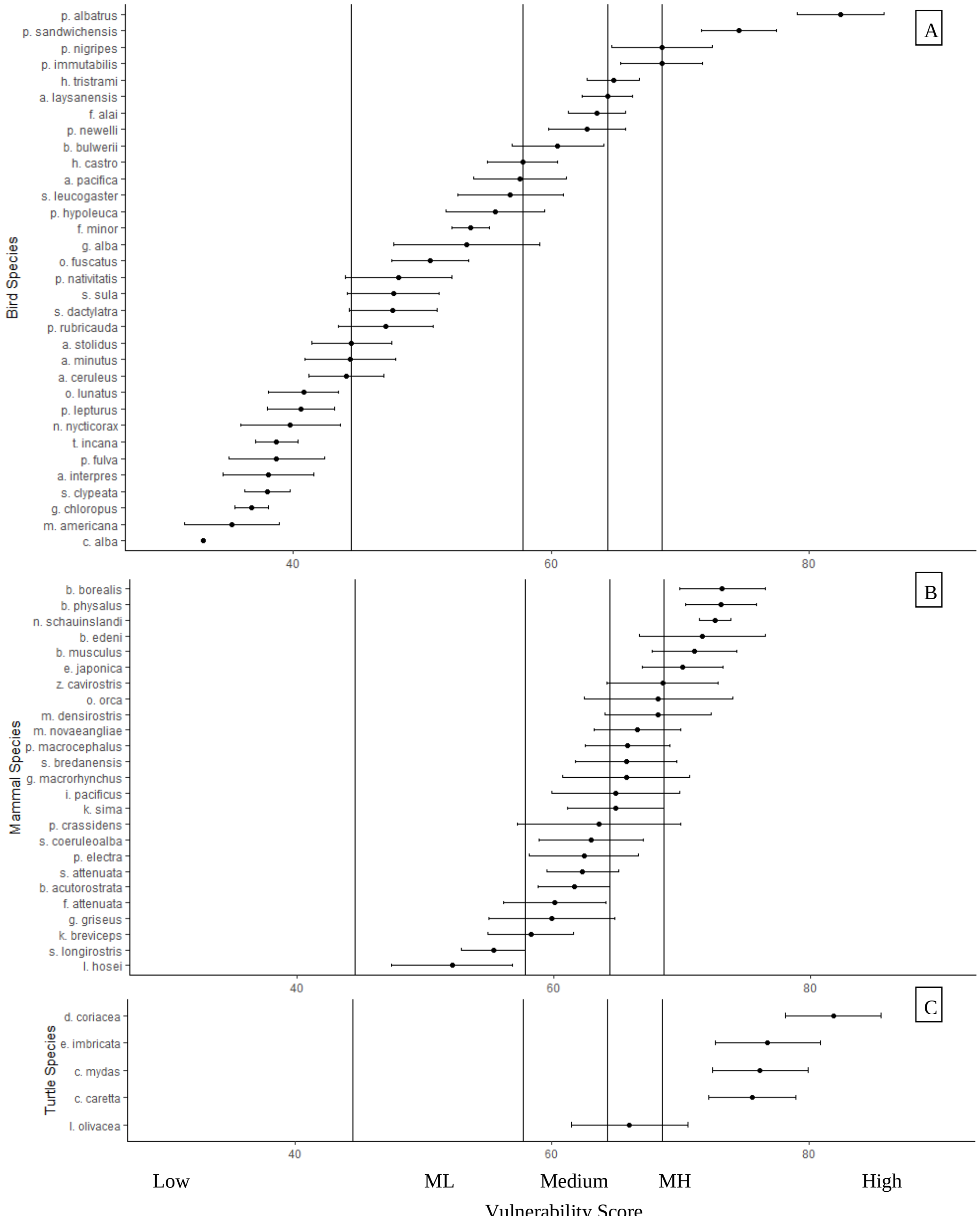

Figure B1. Confidence Intervals for Rankings in the 2.5% SE Scenario. (A) Score with uncertinaty for birds, (B) score with uncertainty for mammals, (C) score with uncertainty for turtles.

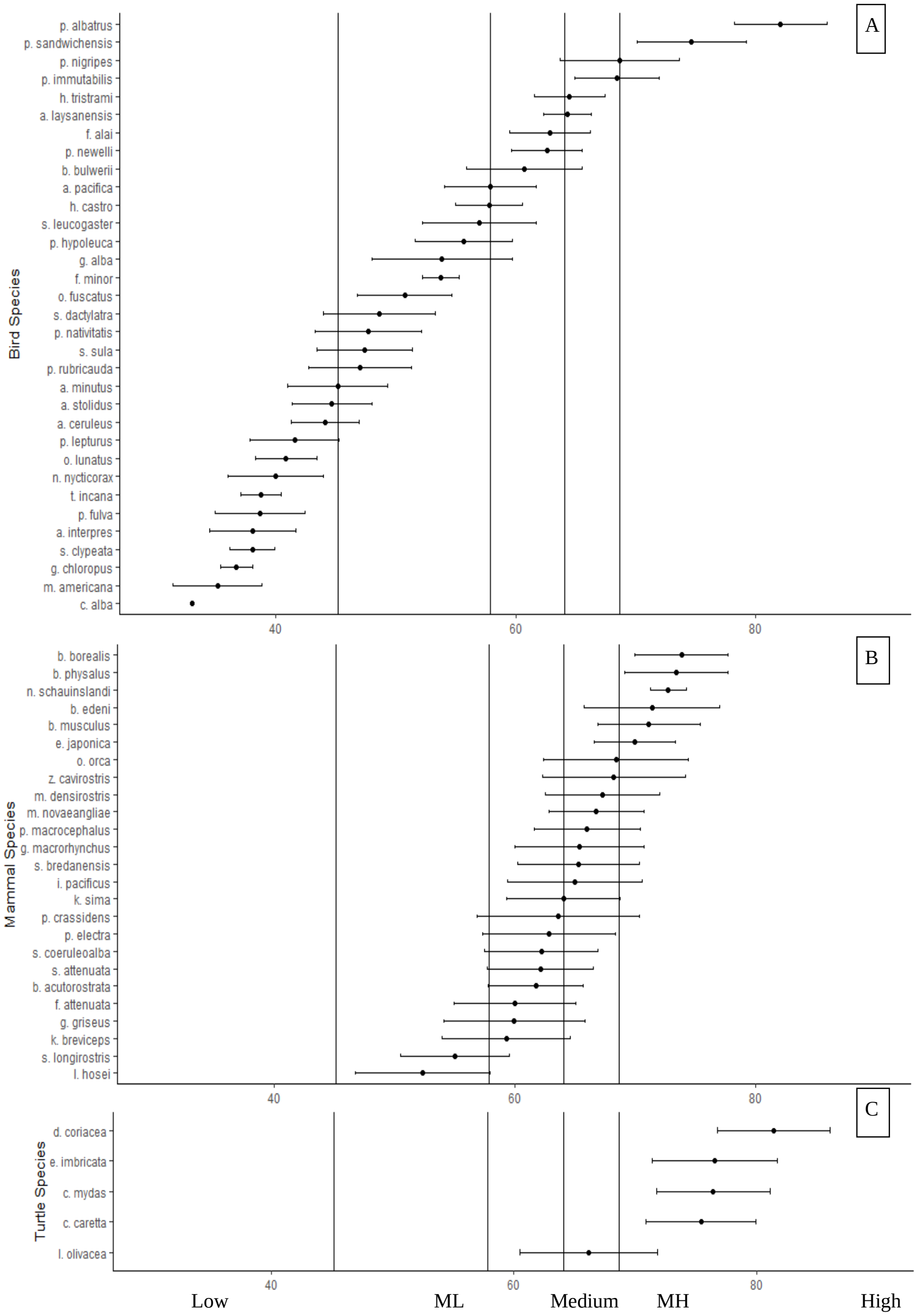

Figure B2. Confidence Intervals for Rankings in the 5% SE Scenario. (A) Score with uncertinaty for birds, (B) score with uncertainty for mammals, (C) score with uncertainty for turtles.

Sensitivity analyses. My tests for correlation between traits showed there was significant correlation between mass and prey preferences, both indicators of species sensitivity, as well as all four population reslience traits. Therefore, I recalculated species rankings without each of these six traits evalute if they were contributing to rankings. All six traits contributed to species rankings with 8 to 28 species switching vulnerability groups, depending on the trait removed. Table C4 shows the final rankings and categories for each species with the removal of each trait. The removal of species sensitivity scores had less influence on the final rankings then the removal of population resilience scores. This was expected, because as more traits are included, the influence of each individula trait will decrease.

Table C4

Species Rankings and Vulnerability Categories with Trait Removal

Trait removed	Generation length	Habitat	Population	IUCN status	Mass	Prey preferences
Low	t. incana	t. incana	t. incana	t. incana	c. alba	c. alba
	c. alba	c. alba	c. alba	c. alba	m. americana	g. chloropus
	p. fulva	p. fulva	s. clypeata	p. fulva	t. incana	m. americana
	p. rubricauda	a. interpres	n. nycticorax	a. laysanensis	a. interpres	p. fulva
	a. interpres	a. ceruleus	p. fulva	a. interpres	g. chloropus	p. lepturus
	s. clypeata	p. rubricauda	a. interpres	s. clypeata	p. fulva	n. nycticorax
	a. ceruleus	p. lepturus	p. lepturus	a. ceruleus	o. lunatus	a. interpres
	n. nycticorax	s. clypeata	a. ceruleus	p. newelli	s. clypeata	o. lunatus
	k. breviceps	m. americana	m. americana	n. nycticorax	p. lepturus	s. clypeata
	l. hosei	o. lunatus	l. hosei	p. rubricauda	n. nycticorax	t. incana
	p. nativitatis	n. nycticorax	o. lunatus	p. lepturus	a. ceruleus	a. minutus
	p. lepturus	a. minutus	a. minutus	k. breviceps	a. minutus	a. ceruleus
Low-Medium	o. lunatus	g. chloropus	g. chloropus	l. hosei	a. stolidus	s. dactylatra
	m. novaeangliae	p. nativitatis	p. rubricauda	m. americana	p. rubricauda	a. stolidus
	i. pacificus	k. breviceps	p. nativitatis	o. lunatus	p. nativitatis	p. rubricauda
	f. attenuata	l. hosei	a. stolidus	a. minutus	s. sula	s. sula
	p. electra	a. stolidus	k. breviceps	p. nativitatis	s. dactylatra	p. nativitatis

	m. americana	a. laysanensis	s. sula	f. alai	o. fuscatus	l. hosei
	a. minutus	g. alba	g. alba	e. japonica	g. alba	o. fuscatus
	p. newelli	s. sula	a. laysanensis	g. chloropus	p. hypoleuca	g. alba
	a. laysanensis	p. newelli	m. novaeangliae	i. pacificus	l. hosei	s. leucogaster
	e. japonica	f. alai	f. attenuata	m. novaeangliae	s. leucogaster	f. minor
	b. acutorostrata	h. tristrami	g. griseus	f. attenuata	f. minor	s. longirostris
	k. sima	s. dactylatra	p. electra	k. sima	s. longirostris	a. pacifica
Medium	g. chloropus	s. leucogaster	s. dactylatra	g. alba	a. pacifica	p. hypoleuca
	g. alba	i. pacificus	p. newelli	b. musculus	h. castro	h. castro
	b. musculus	m. novaeangliae	k. sima	s. sula	b. bulwerii	f. attenuata
	s. sula	f. attenuata	i. pacificus	a. stolidus	f. attenuata	k. breviceps
	a. stolidus	k. sima	b. acutorostrata	p. electra	p. newelli	g. griseus
	g. griseus	f. minor	s. longirostris	h. tristrami	k. breviceps	s. attenuata
	s. bredanensis	h. castro	h. castro	b. acutorostrata	s. attenuata	b. bulwerii
	h. castro	p. electra	p. hypoleuca	g. griseus	g. griseus	p. electra
	h. tristrami	p. hypoleuca	f. alai	h. castro	a. laysanensis	s. bredanensis
	s. dactylatra	s. longirostris	s. leucogaster	s. dactylatra	f. alai	p. crassidens
	f. alai	o. fuscatus	s. attenuata	s. leucogaster	p. electra	p. newelli
	z. cavirostris	e. japonica	f. minor	b. borealis	s. bredanensis	b. acutorostrata
	s. leucogaster	b. acutorostrata	s. bredanensis	s. longirostris	s. coeruleoalba	i. pacificus
Medium -High	f. minor	g. griseus	h. tristrami	f. minor	b. acutorostrata	g. macrorhynchus
	p. crassidens	b. musculus	o. fuscatus	l. olivacea	h. tristrami	s. coeruleoalba
	b. physalus	s. bredanensis	e. japonica	s. bredanensis	m. novaeangliae	p. macrocephalus
	p. hypoleuca	b. bulwerii	s. coeruleoalba	b. physalus	i. pacificus	k. sima
	s. attenuata	p. crassidens	l. olivacea	p. hypoleuca	g. macrorhynchus	f. alai
	s. longirostris	a. pacifica	g. macrorhynchus	n. schauinslandi	p. crassidens	z. cavirostris
	m. densirostris	z. cavirostris	z. cavirostris	b. edeni	k. sima	h. tristrami
	g. macrorhynchus	n. schauinslandi	b. musculus	p. sandwichensis	l. olivacea	m. novaeangliae
	o. orca	s. attenuata	m. densirostris	p. crassidens	p. immutabilis	o. orca
	b. borealis	p. sandwichensis	a. pacifica	z. cavirostris	p. macrocephalus	p. nigripes
	s. coeruleoalba	m. densirostris	b. edeni	o. orca	z. cavirostris	p. immutabilis

	l. olivacea	l. olivacea	p. crassidens	s. attenuata	p. nigripes	m. densirostris
	b. edeni	b. physalus	b. physalus	m. densirostris	m. densirostris	a. laysanensis
High	b. bulwerii	g. macrorhynchus	b. bulwerii	g. macrorhynchus	o. orca	b. edeni
	o. fuscatus	o. orca	b. borealis	o. fuscatus	b. edeni	b. borealis
	p. sandwichensis	b. borealis	p. macrocephalus	s. coeruleoalba	e. japonica	l. olivacea
	a. pacifica	s. coeruleoalba	o. orca	a. pacifica	n. schauinslandi	n. schauinslandi
	p. macrocephalus	b. edeni	n. schauinslandi	b. bulwerii	b. borealis	b. physalus
	n. schauinslandi	c. caretta	p. sandwichensis	p. macrocephalus	b. musculus	e. japonica
	d. coriacea	p. nigripes	d. coriacea	e. imbricata	b. physalus	c. mydas
	c. caretta	e. imbricata	c. mydas	d. coriacea	p. sandwichensis	b. musculus
	p. albatrus	p. albatrus	c. caretta	c. mydas	c. mydas	p. sandwichensis
	e. imbricata	p. macrocephalus	p. nigripes	c. caretta	e. imbricata	c. caretta
	c. mydas	c. mydas	e. imbricata	p. albatrus	c. caretta	e. imbricata
	p. nigripes	d. coriacea	p. immutabilis	p. nigripes	p. albatrus	d. coriacea
	p. immutabilis	p. immutabilis	p. albatrus	p. immutabilis	d. coriacea	p. albatrus

APPENDIX C

TRAIT DATA TO INFORM VULNERABILTY INDEX

Table C1

Exposure Trait Data

species	dist_m	dist_sd	span_m	span_sd
l_olivacea	2.58	0	40	5
e_imbricata	2.62	0	55	2.5
d_coriacea	2.65	0	47.5	1.1875
c_caretta	2.63	0	87.5	8.75
c_mydas	2.54	0	87.5	8.75
m_americana	0.088841481	0	17.5	1.75
a_laysanensis	2.916	0	12	1.2
s_clypeata	0.07276	0	13	1.3
f_alai	2.6877	0	15.5	0.3875
t_incana	0.2172	0	6	0.6
a_interpres	0.3744	0	16.4	1.64
c_alba	0.3933	0	10.9	1.09
p_fulva	0.4578	0	17.5	1.75
o_lunatus	1.8816	0	21.6	2.16
o_fuscatus	2.475	0	30.5	2.25
a_stolidus	2.325	0	21	2
a_minutus	2.052	0	20	2.5
g_alba	2.0868	0	26	5
p_rubricauda	2.346	0	24.5	4.25
p_lepturus	2.4206	0	12	2
s_sula	2.2325	0	31.5	4.25
s_leucogaster	2.4651	0	40	5
n_nycticorax	0.09045	0	17.5	1.75
f_minor	2.223	0	38.5	0.75
p_hypoleuca	3.03	0	23	4
p_sandwichensis	2.5839	0	32.5	1.25
b_bulwerii	2.5245	0	20	2
a_pacifica	2.3343	0	36	3.6
p_nativitatis	2.1146	0	17	3
p_albatrus	2.78	0	42	6.5
p_nigripes	2.4794	0	50	5
p_immutabilis	2.78	0	42	6.5
h_tristrami	2.7455	0	12.5	1.25
a_ceruleus	1.7848	0	15	1.5
s_dactylatra	2.3958	0	23	2.3
g_chloropus	0.09245	0	10.75	0.125
h_castro	2.2477	0	30	3

p_newelli	1.8915	0	30	3
b_acutorostrata	2.47	0	47.5	1.25
b_borealis	2.44	0	72	1
b_edeni	2.63	0	61.00	5.50
b_musculus	2.47	0	85	2.5
b_physalus	2.52	0	95.5	10.25
e_japonica	2.1	0	60	2.5
f_attenuata	2.62	0	20	0.5
g_macrorhynchus	2.65	0	54.5	4.25
g_griseus	2.62	0	25	2.5
i_pacificus	2.54	0	33	3
k_breviceps	2.67	0	17	0
k_sima	2.67	0	19.5	1.25
l_hosei	2.53	0	17	0.5
m_novaeangliae	2.46	0	86	4.5
m_densirostris	2.67	0	27	0
n_schauinslandi	3.17	0	27.5	1.25
o_orca	2.47	0	63	13.5
p_electra	2.52	0	37.5	3.75
p_macrocephalus	2.47	0	77	0
p_crassidens	2.55	0	41	9.5
s_attenuata	2.61	0	46	0
s_coeruleoalba	2.68	0	55	2.5
s_longirostris	2.6	0	26	0
s_bredanensis	2.63	0	40	4
z_cavirostris	2.59	0	36	0

Table C2

Species' Sensitivity Trait Data

species	mass_m	mass_sd	forage	prey	regurgitation	stress_m	stress_sd
l_olivacea	40.00	4.00	1	2	5	31	0
e_imbricata	81.35	22.825	1	2	5	37	0
d_coriacea	575	162.5	3	5	5	22	0
c_caretta	311	117	4	2	5	18	0
c_mydas	147	17	5	4	5	6	0

m_americana	0.998	0.168	3	2	5	13.4	1.34
a_laysanensis	0.25	0.08	3	1	5	14	0
s_clypeata	0.735	0.1325	4	2	5	19	0
f_alai	0.675	0.1125	3	3	5	22	0
t_incana	0.108	0.0035	1	1	2	8	0
a_interpres	0.137	0.0265	1	2	5	12	0
c_alba	0.07	0.015	1	2	1	8	0
p_fulva	0.168	0.03	1	3	4	8	0
o_lunatus	0.133	0.0065	3	3	3	6	0
o_fuscatus	0.193	0.0075	4	4	1	17	0
a_stolidus	0.186	0.0095	4	3	3	9	0
a_minutus	0.1125	0.01375	4	4	2	13	0
g_alba	0.13	0.0065	3	4	5	10	0
p_rubricauda	0.67	0.0125	2	3	3	5	0
p_lepturus	0.36	0.016	1	3	3	14	0
s_sula	0.95	0.025	3	4	3	4	0
s_leucogaster	1.375	0.2125	2	4	3	12	0
n_nycticorax	0.8	0	2	3	3	13.4	1.34
f_minor	1.4	0.2	3	4	3	8	0
p_hypoleuca	0.179	0.0015	1	3	3	8	0
p_sandwichensis	0.415	0.0075	4	4	3	45	0
b_bulwerii	0.1	0.001	5	4	3	24	0
a_pacifica	0.42	0.016	4	5	3	20	0

p_nativitatis	0.34	0.008	4	4	3	2	0
p_albatrus	4.308	0.0515	5	4	5	12	0
p_nigripes	3.4	0.1125	5	5	3	21	0
p_immutabilis	2.5	0.3	5	4	2	15	0
h_tristrami	0.087	0.0015	5	4	4	10	0
a_ceruleus	0.053	0	4	3	3	13.4	1.34
s_dactylatra	1.79	0.285	2	5	3	13	0
g_chloropus	0.3	0	3	4	4	13.4	1.34
h_castro	0.045	0.001	4	5	3	9	0
p_newelli	0.35	0	1	4	3	38	0
b_acutorostrata	7536	768	5	3	3	5	0
b_borealis	32679.5	6339.75	5	4	3	7	0
b_edeni	28411.5	6205.75	5	3	3	6	0
b_musculus	170000	10000	5	1	3	4	0
b_physalus	54431	9072	5	3	3	6	0
e_japonica	77109	6804.5	5	1	3	13	0
f_attenuata	167.5	28.75	2	5	3	6	0
g_macrorhynchus	1996	499	2	4	3	7	0
g_griseus	400	50	2	4	3	17	0
i_pacificus	11500	1150	2	4	3	20	0
k_breviceps	386	34	3	4	3	7	0
k_sima	203.5	34.25	3	4	3	20	0
l_hosei	181.5	11.25	2	4	3	11	0
m_novaeangliae	34500	3250	5	1	3	13	0
m_densirostris	929.5	56.75	3	4	3	20	0
n_schauinslandi	221	25.5	3	4	3	24	0

o_orca	6983.25	1497.875	1	4	3	19	0
p_electra	251.5	11.75	2	4	3	7	0
p_macrocephalus	27215.4	6803.8	2	5	3	9	0
p_crassidens	1378.925	231.3325	1	5	3	6	0
s_attenuata	112.5	26.25	2	4	3	3	0
s_coeruleoalba	154.5	2.25	1	3	3	6	0
s_longirostris	51.5	14.25	1	4	3	12	0
s_bredanensis	125	17.5	1	5	3	5	0
z_cavirostris	2449	317.5	3	4	3	20	0

Table C3

Population Resilience Trait Data

species	pop_m	pop_sd	gen_m	gen_sd	hab_m	hab_sd	iucn_m	iucn_sd
l_olivacea	1,454,500	0	20	0	3	0	3	0
e_imbricata	30,572	1,066	41.75	1.625	11	0	5	0
d_coriacea	70,700	1,684	26	2	3	0	3	0
c_caretta	67,500	3,750	45	0	11	0	3	0
c_mydas	112,500	3,125	44.5	2.5	5	0	4	0
m_americana	2,700,000	0	3.99	0	12	0	1	0
a_laysanensis	590	45	6.5	0	4	0	5	0
s_clypeata	4500000	100000	6.5	0	9	0	1	0
f_alai	2000	700	7	0	7	0	3	0
t_incana	11850	2575	5.7	0	4	0	1	0
a_interpres	400000	50000	7.3	0	14	0	1	0
c_alba	660,000	20,000	8.1	0	8	0	1	0
p_fulva	220,000	15,000	5.6	0	11	0	1	0

o_lunatus	550,000	225,000	12	0	12	0	1	0
o_fuscatus	23000000	0	11.1	0	14	0	1	0
a_stolidus	11000000	5100000	10.1	0	11	0	1	0
a_minutus	1300000	0	8.1	0	10	0	1	0
g_alba	625000	237500	15.3	0	9	0	1	0
p_rubricauda	70000	0	11.7	0	10	0	1	0
p_lepturus	400000	0	8.4	0	10	0	1	0
s_sula	1400000	0	9.42	0	9	0	1	0
s_leucogaster	210526	0	17.3	0	10	0	1	0
n_nycticorax	2150000	790000	8.8	0	6	0	1	0
f_minor	120000	0	16.6	0	11	0	1	0
p_hypoleuca	1,000,000	0	15.6	0	10	0	1	0
p_sandwichensis	12050	2275	19.8	0	5	0	4	0
b_bulwerii	750000	125000	23.9	0	13	0	1	0
a_pacifica	5473684	136842.1	16.5	0	10	0	1	0
p_nativitatis	150000	0	18.3	0	9	0	1	0
p_albatrus	1734	0	24.1	0	5	0	3	0
p_nigripes	139800	0	22	0	14	0	2	0
p_immutabilis	1600000	0	28.5	0	14	0	2	0
h_tristrami	20000	0	14.6	0	9	0	1	0
a_ceruleus	unknown		9.1	0	10	0	1	0
s_dactylatra	fairly common		16.3	0	11	0	1	0
g_chloropus	4550000	825000	5.9	0	15	0	1	0
h_castro	150000	0	19.1	0	9	0	1	0

p_newelli	15000	2500	15.5	0	4	0	5	0
b_acutorostrata	200000	0	22	0	2	0	1	0
b_borealis	50000	0	23.3	0	2	0	4	0
b_edeni	unknown		18	0	2	0	3	1
b_musculus	10,000	2,500	30.8	0	3	0	4	0
b_physalus	100000	0	25.9	0	2	0	3	0
e_japonica	225	12.5	23	0	2	0	4	0
f_attenuata	40125	0	21.5	2.15	3	0	1	0
g_macrorhynchus	700000	0	23.5	0	3	0	1	0
g_griseus	400000	40000	19.6	0	3	0	1	0
i_pacificus	766	76.6	21.5	2.15	3	0	1	0
k_breviceps	rare		12.1	0	3	0	1	0
k_sima	18750	1875	11.7	0	3	0	1	0
l_hosei	375000	37500	11.1	0	2	0	1	0
m_novaeangliae	84000	0	21.5	0	2	0	1	0
m_densirostris	unknown		21.5	2.15	3	0	1	0
n_schauinslandi	632	0	15	0	6	0	4	0
o_orca	50000	0	25.7	0	4	0	3	1
p_electra	75000	7500	23.5	0	3	0	1	0
p_macrocephalus	1500000	0	32	0	3	0	3	0
p_crassidens	40000	4000	25	0	4	0	2	0
s_attenuata	2200000	0	23	0	2	0	1	0
s_coeruleoalba	2105263.158	52631.5789	22.5	0	3	0	1	0
s_longirostris	562500	56250	13.7	0	5	0	1	0
s_bredanensis	196125	19612.5	21	0	2	0	1	0
z_cavirostris	1250000	12500	28.75	1.125	3	0	1	0

APPENDIX D

R SCRIPT TO CALCULATE VULNERABILTY INDEX

R CODE FOR CALCULATING VULNERABILITY INDEX WITH 0% SE

```
library(readxl)
library(ggplot2)
library(tidyverse)
library(dplyr)
library(corrplot)
Framework_SA_R <- read.csv("~/Projects/Impacts/Manuscript 2/DF/Framework_SA-0.csv") #read in data frame that includes trait data for 11 trait indicators for 63 species with standard deviation around trait data provided

#calculating species vulnerability scores
#set seed to conduct 1000 random draws for species trait data based on mean and SD
set.seed(1234) # Set RNG seed
n_rand <- 1000 # Number of random draws
n_spec <- 63 # Number of species

df <- Framework_SA_R #rename data frame
df$pop_m <- as.numeric(gsub(",","",df$pop_m)) # Take out commas in pop_m
df$pop_sd <- as.numeric(gsub(",","",df$pop_sd)) # Take out commas in pop_sd

names(df) <- c("species", names(df[-1]))
```

```
out <- select(df[rep(seq_len(nrow(df)), n_rand), ], species, Taxa, order, family, prey, forage, regurgitation) #create data frame with 1000 replicates for each values of the included variables

#add new column for the 8 indicators that have uncertainty around their values. Includes 1000 draws for each species-trait combo, assuming normal distribution around mean and SD in data frame
out$dist <- rnorm(nrow(out), as.numeric(df$dist_m), as.numeric(df$dist_sd))
out$span <- rnorm(nrow(out), as.numeric(df$span_m), as.numeric(df$span_sd))
out$mass <- rnorm(nrow(out), as.numeric(df$mass_m), as.numeric(df$mass_sd))
out$stress <- rnorm(nrow(out), as.numeric(df$stress_m), as.numeric(df$stress_sd))
out$pop <- rnorm(nrow(out), as.numeric(df$pop_m), as.numeric(df$pop_sd))
out$gen <- rnorm(nrow(out), as.numeric(df$gen_m), as.numeric(df$gen_sd))
out$hab <- rnorm(nrow(out), as.numeric(df$hab_m), as.numeric(df$hab_sd))
out$risk <- rnorm(nrow(out), as.numeric(df$iucn_m), as.numeric(df$iucn_sd))

out <- out[order(out$species), ]

# Can use this to test if your means and standard devs by species look right
#test <- out %>% group_by(species) %>% summarise(mean = mean(dist), sd = sd(dist))

# Come up with quintiles for each column to convert indicators into species scores
out <- out %>%
```

```
  mutate(
    dist_q = cut(out$dist, breaks=quantile(out$dist, probs = seq(0, 1, by = 0.20), na.rm =
TRUE), labels=c(1,2,3,4,5), include.lowest=TRUE),
    span_q = cut(out$span, breaks=quantile(out$span, probs = seq(0, 1, by = 0.20), na.rm
= TRUE), labels=c(1,2,3,4,5), include.lowest=TRUE),
    mass_q = cut(out$mass, breaks=quantile(out$mass, probs = seq(0, 1, by = 0.20), na.rm
= TRUE), labels=c(5,4,3,2,1), include.lowest=TRUE),
    stress_q = cut(out$stress, breaks=quantile(out$stress, probs = seq(0, 1, by = 0.20),
na.rm = TRUE), labels=c(1,2,3,4,5), include.lowest=TRUE),
    pop_q = cut(out$pop, breaks=quantile(out$pop, probs = seq(0, 1, by = 0.20), na.rm =
TRUE), labels=c(5,4,3,2,1), include.lowest=TRUE),
    gen_q = cut(out$gen, breaks=quantile(out$gen, probs = seq(0, 1, by = 0.20), na.rm =
TRUE), labels=c(1,2,3,4,5), include.lowest=TRUE),
    hab_q = cut(out$hab, breaks=quantile(out$hab, probs = seq(0, 1, by = 0.20), na.rm =
TRUE), labels=c(5,4,3,2,1), include.lowest=TRUE)
  )

# Fill in NA values for missing populations
unknown_list <- c("a. ceruleus","b. edeni","m. densirostris")
fc_list <- c("s. dactylatra")
rare_list <- c("k. breviceps")

out$pop_q[out$species %in% unknown_list] <- 3
```

```
out$pop_q[out$species %in% fc_list] <- 2
out$pop_q[out$species %in% rare_list] <- 5

# Make sure everything in "out" is numeric
out <- cbind(out[1], out[2], out[3], out[4],
as.data.frame(lapply(select(out,!c("species","Taxa", "order", "family")), as.numeric)))
# Make sure that the 5-1 quintiles are correct after the switch from the label to the
numeric values
out$mass_q <- (out$mass_q - 6)*(-1)
out$pop_q <- (out$pop_q - 6)*(-1)
out$hab_q <- (out$hab_q - 6)*(-1)
out$exposure = (out$dist_q + out$span_q)/2 *20
out$sensitivity = (out$prey + out$forage + out$regurgitation + out$stress_q +
out$mass_q)/5 * 20
out$resilience = (out$pop_q + out$gen_q + out$risk + out$hab_q)/4 *20
out$total = ((out$dist_q + out$span_q)/2 + (out$prey + out$forage + out$regurgitation +
out$stress_q + out$mass_q)/5 + (out$pop_q + out$gen_q + out$risk + out$hab_q)/4)/3 *
20

# Come up with means and standard devs. of total for each species
out <- out %>%
  group_by(species) %>%
  mutate(total_m = mean(total),
```

```
        exposure_m = mean(exposure),
        sensitivity_m = mean(sensitivity),
        resilience_m = mean(resilience),
        total_sd = sd(total))

out$total_q <- cut(out$total_m, breaks=quantile(out$total_m, probs = seq(0, 1, by = 0.20), na.rm = TRUE), labels=c(1,2,3,4,5), include.lowest=TRUE)

# Remove duplicates and keep only totals, order by ranking, and save as CSV
final <- select(out, species, exposure_m, sensitivity_m, resilience_m, total_m, total_sd, total_q, Taxa, order, family) %>% group_by(species) %>% slice(1) %>% ungroup()
final <- final[order(final$total_m), ]
write.csv(final,"~/Projects/Impacts/Manuscript 2/Final_ranking_0.csv", row.names = FALSE)

# Find final vulnerability scores quintile cutoff values
cutoffs <- quantile(out$total_m, probs = seq(0, 1, by = 0.20), na.rm = TRUE)
cutoffs

#Figure creation
#Subset data by taxa
bird_out <- out[out$Taxa=="bird", c("species", "total", "total_m")]
mammal_out <- out[out$Taxa=="mammal", c("species", "total", "total_m")]
```

```
turtle_out <- out[out$Taxa=="turtle", c("species", "total", "total_m")]


#add total_se to final

final$se = final$total_sd*2


#point plots

#Subset data by taxa

final_bird <- final[final$Taxa=="bird", c("species", "se", "total_m")]

final_mammal <- final[final$Taxa=="mammal", c("species", "se", "total_m")]

final_turtle <- final[final$Taxa=="turtle", c("species", "se", "total_m")]


#point plot for birds. Make sure yintercept lines reflect cutoffs

bird_pplot <- ggplot(final_bird, aes(x=reorder(species, total_m), y=total_m)) +

  geom_point()+

  theme_classic()+

  coord_flip() +

  geom_hline(yintercept = 45.44) +

  geom_hline(yintercept = 58.58) +

  geom_hline(yintercept = 64.52) +

  geom_hline(yintercept = 69.88) +

  labs(x= "Bird Species", y= "Vulnerability Rank Score") +

  geom_errorbar(aes(ymin=total_m-se, ymax=total_m+se), width=.2,

position=position_dodge(0.05))
```

```
bird_pplot

#point plot for mammals
mammal_pplot <- ggplot(final_mammal, aes(x=reorder(species, total_m), y=total_m)) +
  geom_point()+
  theme_classic()+
  coord_flip() +
  geom_hline(yintercept = 45.44) +
  geom_hline(yintercept = 58.58) +
  geom_hline(yintercept = 64.52) +
  geom_hline(yintercept = 69.88) +
  labs(x= "Mammal Species", y= "Vulnerability Rank Score") +
  geom_errorbar(aes(ymin=total_m-se, ymax=total_m+se), width=.2,
position=position_dodge(0.05))
mammal_pplot

#point plot for turtle
turtle_pplot <- ggplot(final_turtle, aes(x=reorder(species, total_m), y=total_m)) +
  geom_point()+
  theme_classic()+
  coord_flip() +
  geom_hline(yintercept = 45.44) +
  geom_hline(yintercept = 58.58) +
```

```
  geom_hline(yintercept = 64.52) +
  geom_hline(yintercept = 69.88) +
  labs(x= "Turtle Species", y= "Vulnerability Rank Score") +
  geom_errorbar(aes(ymin=total_m-se, ymax=total_m+se), width=.2,
position=position_dodge(0.05))
turtle_pplot

#bar graph of count for final score, organized by taxa
ggplot(final, aes(x = total_m, fill = Taxa)) +
  geom_bar(stat = "bin", binwidth = 3) +
  labs(x= "Vulnerability Rank Score", y= "Species count") +
  theme(
    panel.grid.major = element_blank(),
    panel.grid.minor = element_blank()
  ) +
  scale_y_continuous(breaks = seq(0, 10, 2)) +
  scale_fill_manual(values = c("#1b9e77", "#d95f02", "#7570b3"))



#bar graph of count for exposure score, organized by taxa
ggplot(final, aes(x = exposure_m, fill = Taxa)) +
  geom_bar(stat = "bin", binwidth = 3) +
  labs(x= "Vulnerability Rank Score", y= "Species count") +
```

```
  scale_fill_manual(values = c("#1b9e77", "#d95f02", "#7570b3"))

#bar graph of count for sensitivity score, organized by taxa
ggplot(final, aes(x = sensitivity_m, fill = Taxa)) +
  geom_bar(stat = "bin", binwidth = 3) +
  labs(x= "Vulnerability Rank Score", y= "Species count") +
  scale_fill_manual(values = c("#1b9e77", "#d95f02", "#7570b3"))

#bar graph of count for resilience score, organized by taxa
ggplot(final, aes(x = resilience_m, fill = Taxa)) +
  geom_bar(stat = "bin", binwidth = 3) +
  labs(x= "Vulnerability Rank Score", y= "Species count") +
  scale_fill_manual(values = c("#1b9e77", "#d95f02", "#7570b3"))

#Subset data by taxa with family
family_bird <- final[final$Taxa=="bird", c("species", "se", "total_m", "family")]
family_mammal <- final[final$Taxa=="mammal", c("species", "se", "total_m",
"family")]
order_bird <- final[final$Taxa=="bird", c("species", "se", "total_m", "order")]
order_mammal <- final[final$Taxa=="mammal", c("species", "se", "total_m", "order")]

#bar graph of count for total score by mammal family
ggplot(family_mammal, aes(x = total_m, fill = family)) +
```

```
  labs(x= "Vulnerability Rank Score", y= "Species count") +
  geom_bar(stat = "bin", binwidth = 3)

#bar graph of count for total score by bird order
ggplot(order_bird, aes(x = total_m, fill = order)) +
  labs(x= "Vulnerability Rank Score", y= "Species count") +
  geom_bar(stat = "bin", binwidth = 5)

#test for correlation between traits
corr_df <- Framework_SA_R[ , c('dist_m', 'span_m', 'mass_m',
'forage','prey','regurgitation', 'stress_m', 'pop_m', 'gen_m', 'hab_m', "iucn_m")]
names(corr_df) <- c("Distribution", "Longevity", "Mass", "Forage", "Prey", "Egestion",
"Stress", "Population", "Gen Length", "Habitat", "IUCN")
corr_df$Population <- as.numeric(gsub(",","",corr_df$Population)) #remove commas

corr_tab <- cor(na.omit(corr_df))

cor.mtest <- function(corr_df, ...) {
  mat <- as.matrix(corr_df)
  n <- ncol(mat)
  p.mat<- matrix(NA, n, n)
  diag(p.mat) <- 0
  for (i in 1:(n - 1)) {
```

```
    for (j in (i + 1):n) {
      tmp <- cor.test(mat[, i], mat[, j], ...)
      p.mat[i, j] <- p.mat[j, i] <- tmp$p.value
    }
  }
  colnames(p.mat) <- rownames(p.mat) <- colnames(mat)
  p.mat
}
# matrix of the p-value of the correlation
p.mat <- cor.mtest(mtcars)
head(p.mat[, 1:5])

p.mat <- cor.mtest(na.omit(corr_tab))
p.mat
write.csv(p.mat,"~/Projects/Impacts/Manuscript 2/corr.csv", row.names = FALSE)

corrplot(corr_tab, type="upper", order="hclust", p.mat = p.mat, sig.level = 0.05,
tl.col="black")
```