ROADS, REPTILES, AND RECOVERY: APPLYING A COLLABORATIVE DECISION-MAKING APPROACH FOR DIAMONDBACK TERRAPIN (*MALACLEMYS TERRAPIN*) CONSERVATION IN GEORGIA

by

BRIAN ANDREW CRAWFORD

(Under the Direction of John C. Maerz)

ABSTRACT

Natural resource managers frequently face difficult decisions for wicked conservation issues that include multiple threats and management options, competing objectives (e.g., biodiversity, human well-being), diverse stakeholder preferences, and uncertainty of decision outcomes. Structured decision making (SDM) has emerged as an effective tool for addressing such contexts, but SDM has not been applied to one widespread conservation issue: roads. Roads pose a substantial challenge as they simultaneously represent a pervasive threat to wildlife populations and infrastructure important for people's daily lives and local economies. We describe the co-development of a multi-objective SDM framework with local stakeholders to identify management decisions using the diamondback terrapin (*Malaclemys terrapin*) as a focal species and the Jekyll Island Causeway (JIC: Jekyll Island, Georgia, USA) as a model system. Stakeholders specified multiple objectives, including maximizing terrapin population persistence, maximizing patron satisfaction, and minimizing project costs. We employed integrated modeling and decision analysis to explicitly link threats and management actions to population and socioeconomic outcomes. Models were informed by empirical work estimating the effects of roadside vegetation on nest predation, effects of current and potential management actions on terrapin survival and population viability, and management preferences of JIC user groups. We found that two management devices deployed on the JIC (roadside barriers and flashing warning signage) significantly reduced road threats, but the terrapin population continued to decline in the absence of additional actions. The best-performing management strategy was robust to differences in stakeholder values and included complementary actions such as roadside barriers, on-road signage, and vegetation and predator removal. This study demonstrates the efficacy of applying SDM to road management contexts and the need for incorporating human dimensions data into SDM to more accurately represent social objective outcomes. Results from flashing signage on the JIC and additional work using social marketing interventions to address diverse human-wildlife conflicts showed that positive conservation outcomes are achievable through changing human behaviors. Ultimately, the approaches employed in this work – understanding species life history and behavior, focusing on multiple threats and objectives, and integrating people into management solutions – should assist managers in reaching defendable decisions for other challenging conservation issues.

INDEX WORDS: Attitudes, Bayesian analysis, behavior interventions, conservation management, decision analysis, diamondback terrapins, *Malaclemys terrapin*, mark-recapture, population viability analysis, roads, structured decision making

ROADS, REPTILES, AND RECOVERY: APPLYING A COLLABORATIVE DECISION-MAKING APPROACH FOR DIAMONDBACK TERRAPIN (*MALACLEMYS TERRAPIN*) CONSERVATION IN GEORGIA

by

BRIAN ANDREW CRAWFORD

B.S., University of Maryland, 2008

M.S., University of Georgia, 2011

A Dissertation Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

DOCTOR OF PHILOSOPHY

ATHENS, GEORGIA

2016

ROADS, REPTILES, AND RECOVERY: APPLYING A COLLABORATIVE DECISION-MAKING APPROACH FOR DIAMONDBACK TERRAPIN (*MALACLEMYS TERRAPIN*) CONSERVATION IN GEORGIA

by

BRIAN ANDREW CRAWFORD

Major Professor:	John C. Maerz
Committee:	Clinton T. Moore
	Nik Heynen
	Terry M. Norton

Electronic Version Approved:

Suzanne Barbour
Dean of the Graduate School
The University of Georgia
May 2016

DEDICATION

To my family, Mitch, Nancy, Allison, Caitlin, Grandpop, and Maeby, for their unconditional love and support. Oh yeah…and to that live oak for sparing my life after my first field season on Jekyll.

ACKNOWLEDGEMENTS

This dissertation represents the culmination of my contributions to larger efforts to conserve diamondback terrapins on the Georgia Coast. Throughout my master's and now doctoral work, the quantity and quality of support I received is staggering and humbling. The road has been long, the scenery beautiful, and I am so grateful for many people I have come to know along the way.

My time at UGA has solidified my personal mentality that other people know more than me, and I should listen and learn from them. I have a long list of mentors to thank, but my deepest appreciation is for two people who, for the last seven years, have been committed to my development and our collective work: Drs. John Maerz and Terry Norton. John and Terry, you have been so generous with your wisdom, eagerly sharing it with countless students, the public, and me. Thank you so much for guiding me through the design and execution of this project, helping me develop as a critical scholar and passionate practitioner, and leading by example in your tireless work ethic, dedication to engaging the general public, and overall excitement for conservation. The burden of mentoring me has not rested on John and Terry alone. I also thank my committee members, Drs. Clint Moore and Nik Heynen, for your constant, kind support and for pushing me to develop a deeper understanding of areas of conservation I never thought I would be excited about but now know their power for addressing many challenges we face in conservation. Lastly, I thank my mentors from Rare, Dr. Amielle DeWan, Kevin Green, and Monica Pearce, for showing me how to make conservation about people and Pride.

Any accomplishments of this project hinged on the tireless work of a small army of staff and volunteers of the Georgia Sea Turtle Center. I cannot thank this group enough for being our full partners in this work, the ones that were saving turtles on the road before I even knew Georgia had a coast, and the ones that will continue to conserve Georgia's wildlife. I thank Michelle Kaylor for developing project protocols, providing constant logistical support, and always pushing me toward the highest research and personal standards. I thank the rest of the GSTC's cast of characters for assistance with logistics and field work: veterinary and husbandry staff (Amy Hupp, Steven Nelson, Rachel Thomas), Americorps terrapin members (Becca Cozad, Simon Dilts, Anthony Gillis, Dan Quinn, Lisa Rodriguez, Priya Subbarayan), and all other staff, volunteers, and citizens that saved terrapins on the road not because it would get them a dissertation but because, for them, it was the right thing to do. I am also especially grateful for Bob and Lois Weitz and Tammy and Jerry Ledford for opening their homes for Maeby and me for two amazing summers on Jekyll.

This work would also have not been possible, or as gratifying, without the involvement of representatives from Jekyll's stakeholder groups. These individuals committed considerable time and energy and worked together constructively to produce the best decisions possible for meeting the needs of Jekyll's people and wildlife. I thank Jones Hooks and Ben Carswell (Jekyll Island State Park Authority), Kimberly Andrews (Georgia Sea Turtle Center), Gene Jarrett and Franklin Duncan (Jekyll Island residents), Marcie Hunter (Golden Isles Convention and Visitors Bureau), Luke Kelly (Jekyll Island business owners and employees), Alice Keyes (One Hundred Miles), Mark Dodd (Georgia Department of Natural Resources), and Troy Pittman and Teresa Scott (Georgia Department of Transportation).

I thank my Athens family, my friends and labmates, for pretty much everything. It has been amazing getting to know all of you and I cannot thank you enough for your kindness, humor, and all of your solicited or unsolicited life advice. Thank you and cheers to the times and collaborations we will have, Kyle Barrett, Kristen Cecala, Jayna DeVore, Andrew Grosse, Lincoln Larson, Anna McKee, Joe Milanovich, Dara Satterfield, Sean Sterrett, Vanessa Kinney Terrell, Kevin Stohlgren, Jennifer Asper Spatz, Kira McEntire, Adam Clause, Heather Abernathy, Abby Sterling, Ashley Raybould, Greg Skupien, and David Zailo, as well as Mark Gasser and Darren Stimpfle for providing occasional field assistance but always tremendous support from afar. Thank you, Athens, for providing a second home with amazing people and establishments that have helped make graduate school relentlessly enjoyable, including such places as Creature Comforts, Terrapin Brewery, Jittery Joe’s, Cali N Tito’s, the Georgia Theatre, and the 40 Watt Club. I also thank the musicians at Redeemer Presbyterian Church for letting me contribute to joyful music every Sunday.

Lastly, I thank my family. Thank you, to my parents (Mitch and Nancy), sisters (Allison and Caitlin), and Grandpop for always encouraging me to ask questions and investigate the natural world and for providing opportunities to do so through books, trips to nature centers or the Southwest, and an incredibly cool video conference call with NSF scientists in Antarctica when I was 5. Thank you for putting up with me bringing home creatures or talking at length about terrapins. You guys are the best, and I love you.

TABLE OF CONTENTS

Page

APPENDICES

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

There is a growing awareness that most problems facing the field of conservation are wicked in nature (Game *et al.* 2014). By “wicked,” I mean issues that involve complex interactions between social and ecological systems, multiple management objectives that conflict or compete for limited resources (e.g., protect biodiversity, promote human well-being), and substantial uncertainty about the current state of the system being considered and its response to potential solutions (Hirsch *et al.* 2011; McShane *et al.* 2011; Converse *et al.* 2013). Thus, many scholars have advocated for holistic, integrative approaches to address wicked conservation issues (Cortner & Moote 1999; Berkes, Colding & Folke 2003; Keough & Blahna 2006; ACSC 2011). These integrative approaches, as discussed in the Advancing Conservation in a Social Context initiative (2011), draw from diverse disciplines to engage multiple ways of knowing and valuing the world to consider complex socioecological issues. In the past, “win-win” approaches have attempted to meet all objectives simultaneously; however, the many pitfalls and few durable solutions associated with this approach have led to the realization that the costs and benefits of any conservation decision are never distributed uniformly across stakeholders, species, or habitats (ACSC 2011; Hirsch *et al.* 2011). The use of integrative approaches should allow conservation practitioners to shift from a “win-win” approach toward ones that explicitly define and recognize the various tradeoffs involved in decision making (Hirsch *et al.* 2011; McShane *et al.* 2011; Vercoe *et al.* 2014). Acknowledging complexity and identifying potential tradeoffs early in the decision making process can foster realistic expectations, develop creative solutions, and facilitate negotiations among multiple stakeholder groups (Hirsch *et al.* 2011; McShane *et al.* 2011). To address emerging challenges in conservation, systematic methods or tools are needed to operationalize integrative approaches and allow decision makers to navigate complex and uncertain issues to produce robust, equitable outcomes.

STRUCTURED DECISION MAKING

Structured decision making (SDM), a form of decision analysis, has emerged in recent decades as a systematic process in which stakeholders engage to address complex conservation issues (Gregory *et al.* 2012). Defined by Keeney (1982), SDM is "a formalization of common sense for decision problems that are too complex for informal use of common sense." SDM is an approach that separates a decision problem into elements that are based on science (estimating consequences, tradeoffs, uncertainty in a system) with those based on values (identifying problems, objectives, and utilities of stakeholders) to identify optimal management decisions (Clemen & Reilly 2013; Conroy & Peterson 2013). Stakeholders collaboratively advance through the steps of SDM by framing the problem, defining relevant objectives, developing management alternatives, predicting consequences to the system, and examining tradeoffs to identify optimal decisions (Figure 1.1: Gregory *et al.* 2012). Recognized strengths of SDM include the following: i) transparency throughout the process that allows a clearer understanding of uncertainty and anticipated tradeoffs between outcomes of competing objectives, ii) inclusivity of stakeholders' knowledges and perspectives at each step, and iii) emphasis on defining objectives before prioritizing management options (Conroy & Peterson 2013). Readers may refer to Clemen and Reilly (2013) for a deeper coverage of SDM, in general, and to Conroy and Peterson (2013) for SDM within the context of natural resource management.

SDM has been effectively applied to many wildlife management contexts that focus primarily on meeting biological (population) objectives and linking these to actions via rigorous ecological models (e.g., Martin *et al.* 2009; Blomquist *et al.* 2010; Converse *et al.* 2013). Within these contexts, the decision makers, stakeholders, and scientists involved in the SDM process have predominantly trained in natural sciences and are affiliated with ecological or natural resource-focused governmental agencies or academic departments. Social objectives (e.g., maximize personal livelihoods, public satisfaction, agency public relations) are less frequently included in decision contexts and, if included, are evaluated using coarse metrics and expert opinion (e.g., Converse *et al.* 2013). Since it is the norm, and not the exception, that conservation issues include social dimensions that will be affected by most decisions (Hirsch *et al.* 2011), SDM may produce more robust decisions by incorporating data and approaches from the fields of human dimensions and social sciences that directly inform social objectives. For example, survey-based human dimensions research can be used to gauge stakeholders' management preferences and improve the estimation of alternatives' consequences on social outcomes such as public satisfaction (e.g., Vaske & Donnelly 2007). Additionally, decision makers or facilitators could employ stakeholder analysis to identify and characterize groups that should be represented in the SDM process (Brugha & Varvasovszky 2000; Glicken 2000). Despite these gaps, SDM has yielded defendable decisions across many conservation contexts, including endangered species management (e.g., Gregory & Long 2009), invasive species management (e.g., Liu *et al.* 2012), reintroduction (e.g., Converse *et al.* 2013), and formation of policy regulating species harvest (e.g., Nichols & Williams 2006). However, limited work has applied formal decision making approaches to another widespread and wicked conservation problem: roads.

ROADS

Roads have become a pervasive fixture on most physical landscapes with ecological implications for terrestrial and semi-aquatic species; thus, roads present a major challenge to wildlife managers. In the conterminous United States alone, 82% of all land area is within 1 km of the nearest roadway (Riitters & Wickham 2003). Although declining species in landscapes fragmented by roads often face several threats simultaneously, failing to address negative impacts directly caused by roads may compromise the ability of any management practice to facilitate recovery and will lead to a waste of resources. However, mitigating road threats may be more costly or less feasible than other alternatives, so additional threats still must be considered in a management decision context. Numerous management actions exist to mitigate road threats (Glista, DeVault & DeWoody 2009), but implementing effective strategies remains challenging. Although many actions have successfully reduced threats (e.g., roadside barriers for wildlife: Aresco 2005), others have yielded limited or uncertain conservation outcomes (e.g., warning signage for drivers: Putman 1997; Sullivan *et al.* 2004). Furthermore, designing and implementing cost-effective actions is difficult since road threats to local species are often spatially or temporally diffuse; however, concentrated patterns of threats (“hot spots” and “hot moments”) can be identified as targets for management in some contexts (Beaudry, Demaynadier & Hunter 2010; Cureton & Deaton 2012; Crawford *et al.* 2014b). Ultimately, pragmatic conservation management of declining species should identify and reduce threats, including road-associated threats, that will most likely hamper population recovery (Heppell, Crowder & Crouse 1996).

As road networks continue their expansion, ecologists have increasingly examined negative effects of roads on wildlife in order to provide more effective mitigation strategies.

These impacts include the destruction of viable habitat, impediment to movement, increased predation by species subsidized by human activities (e.g., raccoons; *Procyon lotor*), alteration of species behavior, and mortality from vehicles (reviewed by Forman & Alexander 1998; Trombulak & Frissell 2000; Forman *et al.* 2003). Frequently, road studies have focused on the most direct and ubiquitous road-associated threat, i.e. wildlife-vehicle collisions (hereafter referred to as road mortality), and have documented its impacts across vertebrate taxa (reviewed by Fahrig & Rytwinski 2009). Life history and behavioral traits have been used to predict which species should be most vulnerable to road mortality. Species that complete extensive overland movements, do not avoid or are attracted to roads, and have lower reproductive rates and longer generation times are expected to be more susceptible because they will encounter roads more frequently and recover less quickly from road mortality (Gibbs & Shriver 2002; Forman *et al.* 2003; Jaeger *et al.* 2005; Rytwinski & Fahrig 2012). Among these species, road mortality can influence population demographics with respect to sex- and stage-specific survival (Haxton 2000; Mumme *et al.* 2000; Ferreras *et al.* 2001; Row, Blouin-Demers & Weatherhead 2007). Occasionally, studies have been able to directly estimate rates of road mortality and project these impacts on population viability (Mumme *et al.* 2000; Row, Blouin-Demers & Weatherhead 2007), and even fewer studies have measured the impacts of multiple threats simultaneously (Crawford *et al.* 2014a). Investigating multiple threats to species inhabiting road-fragmented landscapes provides a model for reducing uncertainty, predicting consequences to population viability, ranking these threats, and making pragmatic management decisions.

In addition to mitigating road threats to wildlife, multiple management objectives arise as roads also represent valuable infrastructure important to people's daily lives and local economies (Fahrig & Rytwinski 2009; Andrews, Nanjappa & Riley 2015). Most roads exist on public lands and are heavily traveled by people representing a variety of demographics, values, and attitudes. Thus, strategies designed to benefit wildlife may impinge upon other social or economic objectives. For example, management actions, such as reducing speed limits, constructing barriers to keep animals from entering roadways, or restricting vehicle access, likely have direct or unforeseen consequences to drivers' happiness or safety that will need to be traded off with wildlife outcomes. Evaluating the acceptability of these tradeoffs will depend on the values and preferences of public user groups (Stout *et al.* 1993; Tarrant, Bright & Ken Cordell 1997; Sullivan & Messmer 2003; Vaske & Donnelly 2007).

Causeways provide a growing number of tourists access to coastal destinations each year (Baird 2009). With a mosaic of developed and protected areas, these destinations offer a variety of amenities from historical (e.g., museums), social (shops, restaurants), recreational (beaches, waterparks, biking trails), and environmental (wildlife viewing areas, nature centers) sources. Increased pressure from tourism opportunities gives incentive for managers to transform coastal areas to maximize these amenities – a process known as the "commodification of place" (King & Stewart 1996). While anthropogenic changes to the landscape present obvious risks to wildlife, they can also disenfranchise local residents (both permanent and seasonal) and employees that may differ from visitors in which amenities are valued (Boucquey *et al.* 2012). Importantly, the degree a person values the advancement of natural resources relative to anthropocentric goals are strong predictors of his or her attitudes and resulting behaviors (e.g., voting, contributing funds) toward specific conservation actions (Fulton, Manfredo & Lipscomb 1996; Vaske & Donnelly 1999). Due to multiple stakeholder groups using coastal roads, conflicts are likely to emerge around management decisions that promote the values and goals of certain parties but not others.

FOCAL SPECIES AND STUDY AREA

This dissertation demonstrates the development and application of an integrative approach to a wicked conservation issue that includes elements of ecological research, management and monitoring, human dimensions research, and stakeholder engagement that culminates in a collaborative decision making process with local stakeholders to identify conservation management decisions. We apply this approach using the diamondback terrapin (*Malaclemys terrapin*) as a focal species and the Jekyll Island Causeway (Jekyll Island, Georgia, USA) as a model system.

The 8.7-km Downing-Musgrove Causeway (aka Jekyll Island Causeway: JIC) is the only road connecting the mainland with Jekyll Island, Glynn County, GA, USA (31.08°N, 81.47°W). The JIC is characterized as a high-speed (89 km/hr [55 mph]) state highway with average annual daily traffic (AADT) of 3,440 vehicles/day that peaks from May through July, corresponding with increased summer tourism (Georgia Department of Transportation 2014). Roadside habitat varies from open areas to densely vegetated hedgerows of cedars (*Juniperus virginiana*) and wax myrtles (*Myrica cerifera*) maintained as a wind break. The JIC bisects a peninsula of salt marsh roughly 32 km^2 in area consisting of a network of intertidal creeks and high marsh dominated by *Spartina spp*. experiencing semidiurnal tides 2–3 m in amplitude (Grosse *et al.* 2011). Several creeks are in proximity to or cross under the causeway, and the edge of the road is regularly within 20 m of the high tide mark of the high marsh.

Diamondback terrapins inhabit salt marshes along the Eastern and Gulf Coasts of the United States – regions experiencing the fastest annual increases in both the densities and traffic loads of roads (Baird 2009). Terrapins are currently listed as state threatened or “of special concern” in numerous states including Georgia (Georgia Department of Natural Resources

2015). Threats to terrapin populations in Georgia, as well as range-wide, include habitat degradation in the forms of development and bulkheading (Roosenburg 1991; Gibbons *et al.* 2001; Roosenburg *et al.* 2014), mortality in crab traps (Dorcas, Willson & Gibbons 2007; Grosse *et al.* 2011; Chambers & Maerz *in press*), excessive predation by subsidized predators (Szerlag-Egger & McRobert 2007; Munscher *et al.* 2012; Crawford *et al.* 2014a; Maerz, Seigel & Crawford *in press*), and vehicle-induced mortality (Wood & Herlands 1997; Szerlag & McRobert 2006; Crawford *et al.* 2014a; Maerz *et al. in press*). Like many turtles, terrapins have habits and life history traits that make them particularly vulnerable to road-associated threats (Gibbs & Shriver 2002; Aresco 2005; Fahrig & Rytwinski 2009; Maerz *et al. in press*). Each summer, females nest on land and show an attraction to open, elevated habitat – the same habitat provided by roadsides in human-developed landscapes (Butler, Heinrich & Seigel 2006; Szerlag-Egger & McRobert 2007). As a long-lived species, terrapin population stability is strongly dependent on high annual adult survivorship, and even low rates (< 3%) of per-capita road mortality along coastal causeways can be sufficiently high to cause population declines (Wood & Herlands 1997; Grosse *et al.* 2011; Crawford *et al.* 2014a). Roadside habitat (e.g., hedges, open areas) used for nesting can also influence nest predation rates as well as hatchling sex ratios that have the potential to contribute to population declines. Like many turtles, terrapins exhibit temperature-dependent sex determination, with cooler incubation temperatures (<28°C) producing predominantly males and warmer temperatures (>30°C) producing predominantly females (Jeyasuria & Place 1997). Nests laid in open habitat tend to have higher success, develop faster, and produce a higher proportion of female hatchlings while those laid under dense hedges exhibit lower success, longer development times, and male-biased hatchling sex ratios (Burger & Montevecchi 1975; Roosenburg & Place 1994).

Representative of many high-traffic coastal areas, the JIC is a regional hot spot of mortality where 100-400 adult female terrapins are struck and killed each year while attempting to cross the road to nest (recorded since 2007: Crawford *et al.* 2014b; Georgia Sea Turtle Center, unpubl. data). Owing to its association with nesting, road mortality is disproportionately female-biased and concentrated within the nesting season (late April to July: Grosse *et al.* 2011; Crawford *et al.* 2014b). Although nocturnal nesting has been documented at some sites (Wood & Herlands 1997), females generally nest diurnally during weather with minimal cloud cover and high daily temperatures (Seigel 1980; Feinberg & Burke 2003) and during higher tide amplitudes (Burger & Montevecchi 1975; Feinberg & Burke 2003). On the JIC, Crawford *et al.* (2014b) found concentrated temporal peaks of terrapin nesting activity and subsequent road mortality that were associated with a three-hour window around the daily scheduled diurnal high tide. Terrapins show a degree of philopatry and nest site fidelity (Sheridan *et al.* 2010) that contribute to spatially concentrated hot spots of nesting activity and road mortality observed on the JIC (Crawford *et al.* 2014b). Based on simplified, stage-based population models, the terrapin population surrounding the JIC was predicted to decline in the absence of management of road-associated threats (Crawford *et al.* 2014a). In order to mitigate these threats and spur terrapin population growth, hot spots and hot moments of road mortality on the JIC should serve as targets for cost-effective management.

DISSERTATION THEMES

The work in this dissertation reflects three broad, interwoven themes important for developing conservation strategies that effectively address the complexity, uncertainty, and contention inherent in wicked conservation issues. These themes are developed through five empirical chapters.

First, ecological research of species demography, life history, and behavior is necessary to develop effective and targeted management strategies. We build on previous research that used patterns of nesting and road-crossing habits and identified discrete hot spots and hot moments for management (Crawford *et al.* 2014a; Crawford *et al.* 2014b). Although road mortality occurs across broader places and periods, these hot spots and hot moments represent priorities for cost-effective management. Chapter 2 focuses on the effects of roadside vegetation on predation rates and hatchling sex ratios of terrapin nests with implications of how nest site choice influences demographic rates. Limited work has estimated these interactions, but vegetation management may be a viable component of more holistic strategies if vegetation impacts demographic rates at levels sufficient to influence population growth and persistence. Chapter 4 uses mark-recapture data to estimate key demographic processes, such as the survival of individuals that are or are not crossing (i.e., exposed to the risk of road mortality) and the proportion of nesting females that cross the road. This chapter predicts population persistence under no management, current strategies, and potential strategies designed to influence different portions of the population. Multiple actions within management strategies have been implemented or proposed to target hot spots and hot moments of road mortality in their designs, which have been based around behavioral patterns. Like other road contexts, predicting the consequences of management on the JIC hinges on a firm understanding of current species behavior and population demography as well as how these are altered by anthropogenic stressors and management.

Second, conservation issues are inherently complex, and effective research and management will require a multi-faceted focus. We focus on multiple road-associated threats operating on Jekyll Island by estimating effects of vegetation on demographic processes related to nesting (Chapter 2) and road mortality on adult female survival (Chapter 4). Since ecological researchers often must estimate population dynamics with partial, incomplete datasets, Chapter 4 focuses on using multiple data sources (e.g., mark-recapture, mark-recapture-recovery, published literature, expert opinion) in integrated models to produce more accurate estimates. Chapter 5 integrates human dimensions data from Chapter 3, population estimates from Chapter 4, and expert opinion to better estimate social and ecological consequences of management actions in a structured decision making framework. Furthermore, the management strategies evaluated in Chapters 4 and 5 each contain multiple actions designed to complementarily address these threats. Lastly, the decision making framework in Chapter 5 considers multiple management objectives that span ecological, social, and economic dimensions. Moving from a singular to a multi-faceted focus will allow for conservation solutions to prioritize threats, actions, and objectives that will optimally mitigate anthropogenic stressors to declining species and satisfy stakeholders, as well as guard against unexpected consequences of failing to consider additional threats, objectives, or stakeholders.

Third, conservation is ultimately about people and their behaviors (Schultz 2011), and effective strategies come from integrating people into the development of conservation solutions instead of simply viewing humans as the source of conservation problems. Chapter 3 assesses the values, attitudes, and management preferences of Jekyll Island’s patrons. This research was essential in anticipating the effects of different actions on people’s satisfaction and reducing areas of contention prior to implementing strategies. Chapter 4 includes an explicit example of involving humans in solutions to human-wildlife conflicts on roads; we implemented flashing warning signage designed to alert drivers to terrapins on the road and estimated changes in terrapin survival. Chapter 5 describes the collaborative decision making process we employed on

Jekyll Island where representatives from multiple stakeholder groups engaged to discuss objectives, develop creative solutions, and agree on optimal management strategies. Lastly, Chapter 6 focuses on work I completed through a conservation fellowship with Rare, an international, non-governmental organization based in Arlington, VA. Rare's approach to conservation is people-focused; practitioners work with local communities to conduct social marketing campaigns (interventions) designed to replace human practices that are destructive to natural resources with sustainable ones (Jenks, Vaughan & Butler 2010). The chapter describes a meta-analysis estimating the effectiveness of this approach for changing a community's pro-conservation knowledge, attitudes, and behaviors. Together, the body of work in this dissertation demonstrates effective approaches for integrating people as active participants in decision making and management interventions that promote equitable conservation outcomes.

In addition to my dissertation research, I have pursued several opportunities in strategic communication, collaborative studies, and public outreach and education that have helped me develop as an agile, integrative scientist. Strategically communicating with technical and nontechnical stakeholders was paramount throughout the SDM process discussed in Chapter 5, and I had the opportunity to develop an additional communication tool for the Southeast Climate Science Center. Along with US Geological Survey scientists and other colleagues, I contributed to a factsheet on scientific uncertainty and decision making for the general public (Appendix E). I have also developed my skills as an effective presenter and communicator through my involvement with numerous scientific communication workshops, both as a leader and an attendee. I have engaged in several collaborative studies outside of the scope of this dissertation to address conservation-relevant issues including students' and professionals' attitudes toward advocacy in science (Crawford, Kramer & Hinton 2016), drivers' bias toward hitting snakes relative to other wildlife taxa (Crawford & Andrews 2016), and patrons' attitudes towards rattlesnakes and overall satisfaction at the Claxton Wildlife Festival (Claxton, GA) to inform future outreach programs for the Center for Snake Conservation and Georgia DNR staff. Public outreach and education has also been a routine part of my graduate activities. I co-developed and implemented a lesson plan on urban watershed conservation for Athens elementary schools, taught lessons on reptiles and amphibians, adaptations, and classification to local primary schools, and presented on terrapin ecology and conservation to students at the Georgia Sea Turtle Center's Sea Turtle Camp. I have valued these outreach opportunities greatly, and I will continue to be committed to this important and fruitful approach for engaging audiences of all ages and fostering knowledge and excitement about conservation. Taken together, these experiences have developed my skills for integrating research of ecological and social dimensions, developing innovative approaches and management actions, and meaningfully connecting different audiences to each other and the conservation issues in which they have a stake. Communication and collaboration have been cornerstones of my dissertation and broader work, and these are invaluable tools for advancing conservation by integrating people into solutions instead of only viewing them as the problem.

REFERENCES

ACSC (2011) *Advancing conservation in a social context: working in a world of trade-offs,* Final Report edn. Available from Available online at: http://cicr.uga.edu/acsc/

Andrews, K.M., Nanjappa, P. & Riley, S.P.D. (2015) *Roads and Ecological Infrastructure: Concepts and Applications for Small Animals*. Johns Hopkins University Press, Baltimore, MD.

Aresco, M.J. (2005) Mitigation measures to reduce highway mortality of turtles and other herpetofauna at a north Florida lake. *Journal of Wildlife Management,* **69,** 549-560.

Baird, R.C. (2009) Coastal urbanization: the challenge of management lag. *Management of Environmental Quality: An International Journal,* **20,** 371-382.

Beaudry, F., Demaynadier, P.G. & Hunter, M.L. (2010) Identifying hot moments in road-mortality risk for freshwater turtles. *Journal of Wildlife Management,* **74,** 152-159.

Berkes, F., Colding, J. & Folke, C. (2003) *Navigating social-ecological systems: building resilience for complexity and change*. Cambridge: Cambridge University Press,

Blomquist, S.M., Johnson, T.D., Smith, D.R., Call, G.P., Miller, B.N., Thurman, W.M., McFadden, J.E., Parkin, M.J. & Boomer, G.S. (2010) Structured decision-making and rapid prototyping to plan a management response to an invasive species. *Journal of Fish and Wildlife Management,* **1,** 19-32.

Boucquey, N., Campbell, L.M., Cumming, G., Meletis, Z.A., Norwood, C. & Stoll, J. (2012) Interpreting amenities, envisioning the future: Common ground and conflict in North Carolina's rural coastal communities. *GeoJournal,* **77,** 83-101.

Brugha, R. & Varvasovszky, Z. (2000) Stakeholder analysis: a review. *Health policy and planning,* **15,** 239-246.

Burger, J. & Montevecchi, W.A. (1975) Nest Site Selection in Terrapin *Malaclemys terrapin*. *Copeia,* **1975,** 113-119.

Butler, J., Heinrich, G. & Seigel, R. (2006) Third workshop on the ecology, status, and conservation of diamondback terrapins (*Malaclemys terrapin*): results and recommendations. *Chelonian Conservation and Biology,* **5,** 331-334.

Chambers, R.M. & Maerz, J.C. (*in press*) Terrapin bycatch in the blue crab fishery. *Ecology and conservation of the diamondback terrapin* (eds W.M. Roosenburg & V.S. Kennedy). Johns Hopkins University Press, Baltimore, MD.

Clemen, R. & Reilly, T. (2013) *Making Hard Decisions with Decision Tools,* Third edn. Cengage Learning, Boston, MA.

Conroy, M.J. & Peterson, J.T. (2013) *Decision making in natural resource management: a structured, adaptive approach*. Wiley,

Converse, S.J., Moore, C.T., Folk, M.J. & Runge, M.C. (2013) A matter of tradeoffs: reintroduction as a multiple objective decision. *The Journal of Wildlife Management,* **77,** 1145-1156.

Cortner, H.J. & Moote, M.A. (1999) *The politics of ecosystem management*. Island Press, Washington, D.C.

Crawford, B. & Andrews, K. (2016) Drivers' attitudes toward wildlife-vehicle collisions with reptiles and other taxa. *Animal Conservation*. doi: 10.1111/acv.12261

Crawford, B. A., Kramer, D. W. & Hinton, J. W. (2016) Comparing Student and Professional Responses Toward Advocacy in Science. *Human Dimensions of Wildlife*. doi: 10.1080/10871209.2016.1149747

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A. & Norton, T.M. (2014a) Estimating the consequences of multiple threats and management strategies for semi-aquatic turtles. *Journal of Applied Ecology*, **51,** 359-366.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A., Norton, T.M. & Albeke, S.E. (2014b) Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology,* **51,** 367-375.

Cureton, J.C. & Deaton, R. (2012) Hot moments and hot spots: identifying factors explaining temporal and spatial variation in turtle road mortality. *Journal of Wildlife Management,* **76,** 1047-1052.

Dorcas, M.E., Willson, J.D. & Gibbons, J.W. (2007) Crab trapping causes population decline and demographic changes in diamondback terrapins over two decades. *Biological Conservation,* **137,** 334-340.

Fahrig, L. & Rytwinski, T. (2009) Effects of roads on animal abundance: an empirical review and synthesis. *Ecology and Society,* **14,** 21.

Feinberg, J.A. & Burke, R.L. (2003) Nesting ecology and predation of diamondback terrapins, *Malaclemys terrapin,* at Gateway National Recreation Area, New York. *Journal of Herpetology,* **37,** 517-526.

Ferreras, P., Gaona, P., Palomares, F. & Delibes, M. (2001) Restore habitat or reduce mortality? Implications from a population viability analysis of the Iberian lynx. *Animal Conservation,* **4,** 265-274.

Forman, R.T., Sperling, D., Bissonette, J.A., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R., Goldman, C.R., Heanue, K., Jones, J.A., Swanson, F.J., Turrentine, T. & Winter, T.C. (2003) *Road ecology: science and solutions*. Island Press, Washington.

Forman, R.T.T. & Alexander, L.E. (1998) Roads and their major ecological effects. *Annual Review of Ecology and Systematics,* **29,** 207-231.

Fulton, D.C., Manfredo, M.J. & Lipscomb, J. (1996) Wildlife value orientations: A conceptual and measurement approach. *Human Dimensions of Wildlife,* **1,** 24-47.

Game, E.T., Meijaard, E., Sheil, D. & McDonald-Madden, E. (2014) Conservation in a Wicked Complex World; Challenges and Solutions. *Conservation Letters,* **7,** 271-277.

Georgia Department of Natural Resources (2015) Georgia State Wildlife Action Plan. Wildlife Resources Division, Available from http://www.gadnr.org/cwcs/Documents/strategy.html (accessed on 15 May 2009).

Georgia Department of Transportation (2014) Georgia's State Traffic and Report Statistics (STARS). Available from http://www.dot.state.ga.us/statistics/stars/Pages/GlynnTraffic.aspx (accessed on 19 Oct 2011).

Gibbons, J.W., Lovich, J.E., Tucker, A.D., Fitzsimmons, N.N. & Greene, J.L. (2001) Demographic and ecological factors affecting conservation and management of the Diamondback Terrapin (*Malaclemys terrapin*) in South Carolina. *Chelonian Conservation and Biology,* **4,** 66-74.

Gibbs, J.P. & Shriver, W.G. (2002) Estimating the effects of road mortality on turtle populations. *Conservation Biology,* **16,** 1647-1652.

Glicken, J. (2000) Getting stakeholder participation ‘right’: a discussion of participatory processes and possible pitfalls. *Environmental Science & Policy,* **3,** 305-310.

Glista, D.J., DeVault, T.L. & DeWoody, J.A. (2009) A review of mitigation measures for reducing wildlife mortality on roadways. *Landscape and Urban Planning,* **91,** 1-7.

Gregory, R., Failing, L., Harstone, M., Long, G., McDaniels, T. & Ohlson, D. (2012) *Structured decision making: a practical guide to environmental management choices*. John Wiley & Sons,

Gregory, R. & Long, G. (2009) Using structured decision making to help implement a precautionary approach to endangered species management. *Risk Analysis,* **29,** 518-532.

Grosse, A.M., Maerz, J.C., Hepinstall-Cymerman, J. & Dorcas, M.E. (2011) Effects of roads and crabbing pressures on diamondback terrapin populations in coastal Georgia. *Journal of Wildlife Management,* **75,** 762-770.

Haxton, T. (2000) Road mortality of snapping turtles, *Chelydra serpentina*, in central Ontario during their nesting period. *Canadian Field-Naturalist,* **114,** 106-110.

Heppell, S.S., Crowder, L.B. & Crouse, D.T. (1996) Models to evaluate headstarting as a management tool for long-lived turtles. *Ecological Applications,* **6,** 556-565.

Hirsch, P.D., Adams, W.M., Brosius, J.P., Zia, A., Bariola, N. & Dammert, J.L. (2011) Acknowledging Conservation Trade-Offs and Embracing Complexity. *Conservation Biology,* **25,** 259-264.

Jaeger, J.A.G., Bowman, J., Brennan, J., Fahrig, L., Bert, D., Bouchard, J., Charbonneau, N., Frank, K., Gruber, B. & von Toschanowitz, K.T. (2005) Predicting when animal populations are at risk from roads: an interactive model of road avoidance behavior. *Ecological Modelling,* **185,** 329-348.

Jenks, B., Vaughan, P.W. & Butler, P.J. (2010) The evolution of Rare Pride: Using evaluation to drive adaptive management in a biodiversity conservation organization. *Evaluation and Program Planning,* **33,** 186-190.

Jeyasuria, P. & Place, A.R. (1997) Temperature-dependent aromatase expression in developing diamondback terrapin (*Malaclemys terrapin*) embryos. *Journal of Steroid Biochemistry and Molecular Biology,* **61,** 415-425.

Keeney, R.L. (1982) Decision analysis: an overview. *Operations research,* **30,** 803-838.

Keough, H.L. & Blahna, D.J. (2006) Achieving integrative, collaborative ecosystem management. *Conservation Biology,* **20,** 1373-1382.

King, D.A. & Stewart, W.P. (1996) Ecotourism and commodification: protecting people and places. *Biodiversity & Conservation,* **5,** 293-305.

Liu, S., Walshe, T., Long, G. & Cook, D. (2012) Evaluation of Potential Responses to Invasive Non-Native Species with Structured Decision Making. *Conservation Biology,* **26,** 539-546.

Maerz, J.C., Seigel, R.A. & Crawford, B.A. (*in press*) Terrapin conservation: Mitigating habitat loss, road mortality, and subsidized predators. *Ecology and conservation of the diamondback terrapin* (eds W.M. Roosenburg & V.S. Kennedy). Johns Hopkins University Press, Baltimore, MD.

Martin, J., Runge, M.C., Nichols, J.D., Lubow, B.C. & Kendall, W.L. (2009) Structured decision making as a conceptual framework to identify thresholds for conservation and management. *Ecological Applications,* **19,** 1079-1090.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L. & Pulgar-Vidal, M. (2011) Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation,* **144,** 966-972.

Mumme, R.L., Schoech, S.J., Woolfenden, G.E. & Fitzpatrick, J.W. (2000) Life and Death in the Fast Lane: Demographic Consequences of Road Mortality in the Florida Scrub-Jay. *Conservation Biology*, **14,** 501-512.

Munscher, E.C., Kuhns, E.H., Cox, C.A. & Butler, J.A. (2012) Decreased nest mortality for the Carolina diamondback terrapin (*Malaclemys terrapin centrata*) following removal of raccoons (*Procyon lotor*) from a nesting beach in northeastern Florida. *Herpetological Conservation and Biology,* **7,** 176-184.

Nichols, J.D. & Williams, B.K. (2006) Monitoring for conservation. *Trends in Ecology & Evolution,* **21,** 668-673.

Putman, R.J. (1997) Deer and road traffic accidents: Options for management. *Journal of Environmental Management,* **51,** 43-57.

Riitters, K.H. & Wickham, J.D. (2003) How far to the nearest road? *Frontiers in Ecology and the Environment,* **1,** 125-129.

Roosenburg, W. M. (1991) The diamondback terrapin: habitat requirements, population dynamics, and opportunities for conservation. Pages 227-234 in J.A. Mihursky and A. Chaney (eds.). New Perspectives in the Chesapeake System: a Research and Management Partnership. Chesapeake Research Consortium Publication Number 137, Solomons, MD.

Roosenburg, W.M. & Place, A.R. (1994) Nest predation and hatchling sex ratio in the diamondback terrapin: implications for management and conservation. *Tortoise Sustainable Coastal Watershed: The Chesapeake Experiment. Chesapeake Research Consortium, Publication***,** 65-70.

Roosenburg, W. M., D. M. Spontak, S. P. Sullivan, E. L. Matthews, M. L. Heckman, R. J. Trimbath, R. P. Dunn, E. A. Dustman, L. Smith and L. J. Graham. (2014) Nesting habitat creation enhances recruitment in a predator--free environment: Malaclemys nesting at the Paul S. Sarbanes Ecosystem Restoration Project. *Restoration Ecology,* **22,** 815-823.

Row, J.R., Blouin-Demers, G. & Weatherhead, P.J. (2007) Demographic effects of road mortality in black ratsnakes (< i> Elaphe obsoleta</i>). *Biological Conservation,* **137,** 117-124.

Rytwinski, T. & Fahrig, L. (2012) Do species life history traits explain population responses to roads? A meta-analysis. *Biological Conservation,* **147,** 87-98.

Schultz, P. (2011) Conservation means behavior. *Conservation Biology,* **25,** 1080-1083.

Seigel, R.A. (1980) Nesting habits of Diamondback Terrapins (*Malaclemys terrapin*) on the Atlantic coast of Florida. *Transactions of the Kansas Academy of Sciences,* **83,** 239-246.

Sheridan, C.M., Spotila, J.R., Bien, W.F. & Avery, H.W. (2010) Sex-biased dispersal and natal philopatry in the diamondback terrapin, Malaclemys terrapin. *Molecular Ecology,* **19,** 5497-5510.

Stout, R.J., Stedman, R.C., Decker, D.J. & Knuth, B.A. (1993) Perceptions of risk from deer-related vehicle accidents: implications for public preferences for deer herd size. *Wildlife Society Bulletin,* **21,** 237-249.

Sullivan, T.L. & Messmer, T.A. (2003) Perceptions of deer-vehicle collision management by state wildlife agency and department of transportation administrators. *Wildlife Society Bulletin***,** 163-173.

Sullivan, T.L., Williams, A.F., Messmer, T.A., Hellinga, L.A. & Kyrychenko, S.Y. (2004) Effectiveness of temporary warning signs in reducing deer-vehicle collisions during mule deer migrations. *Wildlife Society Bulletin,* **32,** 907-915.

Szerlag-Egger, S. & McRobert, S.P. (2007) Northern diamondback terrapin occurrence, movement, and nesting activity along a salt marsh access road. *Chelonian Conservation and Biology*, **6,** 295-301.

Szerlag, S. & McRobert, S.P. (2006) Road occurrence and mortality of the northern diamondback terrapin, *Malaclemys terrapin terrapin,* during the nesting season. *Applied Herpetology,* **3,** 27-37.

Tarrant, M.A., Bright, A.D. & Ken Cordell, H. (1997) Attitudes toward wildlife species protection: Assessing moderating and mediating effects in the value-attitude relationship. *Human Dimensions of Wildlife,* **2,** 1-20.

Trombulak, S.C. & Frissell, C.A. (2000) Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*, **14,** 18-30.

Vaske, J.J. & Donnelly, M.P. (1999) A value-attitude-behavior model predicting wildland preservation voting intentions. *Society & Natural Resources,* **12,** 523-537.

Vaske, J.J. & Donnelly, M.P. (2007) *Public knowledge and perceptions of the desert tortoise*. Colorado State University, Fort Collins, CO.

Vercoe, R.A., Welch-Devine, M., Hardy, D., Demoss, J.A., Bonney, S.N., Allen, K., Brosius, P., Charles, D., Crawford, B., Heisel, S., Heynen, N., de Jesús-Crespo, R.G., Nibbelink, N., Parker, L., Pringle, C., Shaw, A. & Van Sant, L. (2014) Acknowledging Trade-offs and Understanding Complexity: Exurbanization Issues in Macon County, North Carolina. *Ecology and Society,* **19,** 23.

Wood, R. & Herlands, R. (1997) Turtles and tires: the impact of roadkills on northern diamondback terrapin, *Malaclemys terrapin terrapin*, populations on the Cape May Peninsula, southern New Jersey, USA. *Conservation, Restoration, and Management of*

Tortoises and Turtles - An International Conference (ed J. Van Abbema), pp. 46-53. New York Turtle and Tortoise Society, New York.

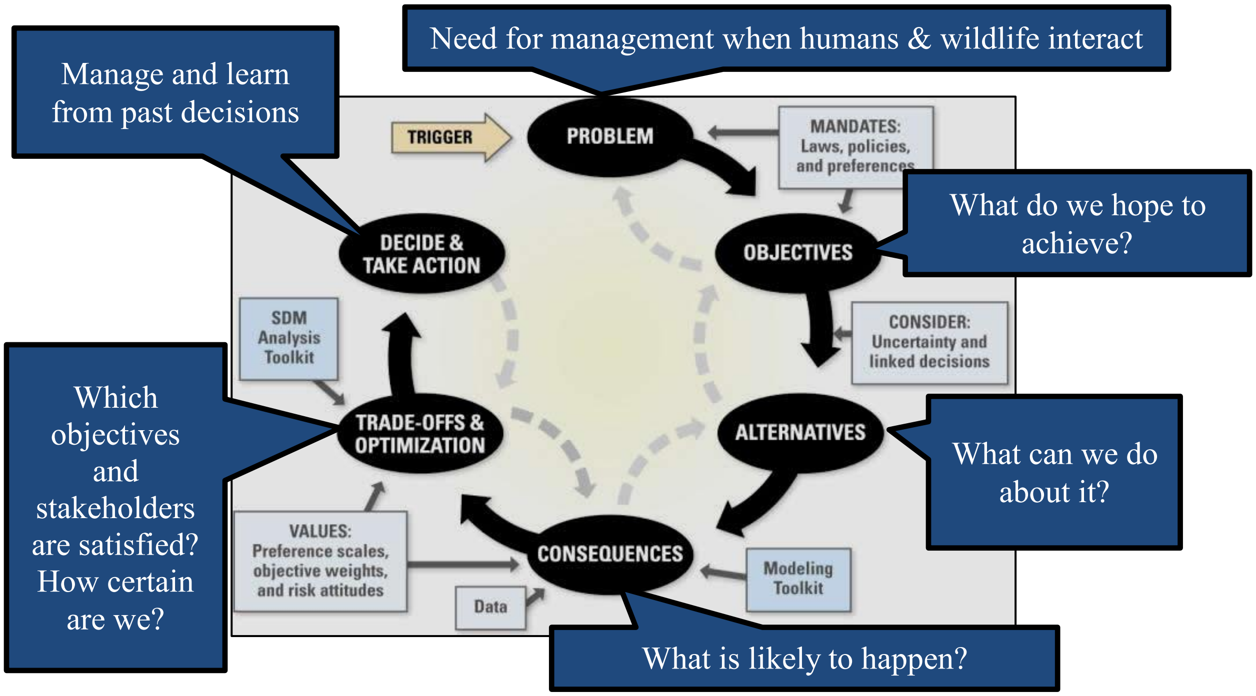


Figure 1.1. Conceptual diagram of iterative structured decision making process showing specific steps and general questions addressed by each concerning conservation issues involving multiple stakeholders, objectives, and choices (alternatives). Modified from figure available at the US Fish & Wildlife Service's National Conservation Training Center website (http://training.fws.gov/).

CHAPTER 2

EFFECTS OF VEGETATION AND ARTIFICIAL NESTING HABITAT MANAGEMENT ON NEST SURVIVAL OF CAROLINA DIAMONDBACK TERRAPINS (*MALACLEMYS TERRAPIN CENTRATA*)

FORWARD

This chapter presents work done in conjunction with a companion study on the effects of roadside vegetation types on hatchling sex ratios of diamondback terrapins, and it is presented in full in Grosse *et al.* (2015).

INTRODUCTION

In many cases there are multiple factors that contribute to wildlife population declines such that management will require simultaneous, integrated interventions to stabilize and recover populations. In the case of diamondback terrapins *Malaclemys terrapin*, populations are perceived to be declining due to multiple factors, including crab-trap mortality (Dorcas *et al.* 2007; Grosse *et al.* 2011), vehicle-induced mortality (Wood & Herlands 1997; Szerlag & McRobert 2006), high nest predation from subsidized and introduced predators (Feinberg & Burke 2003; Szerlag & McRobert 2006), and habitat degradation and alteration (Seigel 1993; Gibbons *et al.* 2001).

Numerous studies have shown that bycatch in crab pots is associated with local and regional declines in terrapin populations (Dorcas *et al.* 2007; Grosse *et al.* 2011; Chambers & Maerz, In review). In addition, terrapin populations are affected by a suite of factors, including high nest depredation and high mortality of gravid females along roads that bisect marsh habitats, both of which can independently and collectively contribute to terrapin population declines (Wood & Herlands 1997; Crawford *et al.* 2014a; Maerz *et al.* In review). Prior management strategies have focused on reducing bycatch rates (Grosse *et al.* 2011; Hart & Crowder 2011; Chambers & Maerz, In review) and road mortality (Crawford *et al.* 2014b). However, addressing those two threats may be insufficient to stabilize or recover some terrapin populations (Crawford *et al.* 2014a). Management interventions to address nest survival and performance are also needed as part of an integrated approach to terrapin management.

Major factors affecting terrapin nest success include loss and degradation of nesting habitats and high nest depredation by subsidized mammalian predators. Nesting habitat may be lost to bulkheading, constructing protective barriers to stabilize the shoreline (Roosenburg 1991, 1994; Winters 2013), and land conversion degraded by soil modification (Roosenburg 1994; Wnek 2010) and the succession of woody vegetation or invasive plants that increase shade (Roosenburg 1991; Wnek 2010). Like many turtles, terrapins exhibit temperature-dependent sex determination, with cooler incubation temperatures (<28°C) producing predominantly males and warmer temperatures (>30°C) producing predominantly females (Jeyasuria & Place 1997). Terrapins prefer to nest in areas of patchy, short vegetation; nests in open habitat develop faster and tend to produce a higher proportion of female hatchlings (Burger & Montevecchi 1975; Goodwin 1994; Roosenburg 1994; Feinberg & Burke 2003; Ner 2003; Scholz 2006; Hackney 2010). Dense vegetation shades nests, resulting in longer development times, higher egg mortality, and male-biased hatchling sex ratios (Wnek 2010). Mortality of terrapins on land, particularly nest and hatchling mortality, is clearly related to subsidized predator abundance. Roosenburg and Place (1995) found that shaded nests in dense grass had higher survival rates but produced almost 100% male hatchlings. Numerous studies show that raccoon *Procyon lotor* predation is the biggest determinant of terrapin nest success, and in many studies at sites with high raccoon densities throughout the species' range, raccoons routinely depredate as many as 95% of nests (Burger 1976, 1977; Roosenburg 1991; Goodwin 1994; Roosenburg & Place 1995; Feinberg & Burke 2003; Ner 2003; Butler *et al.* 2006; Munscher *et al.* 2012; Crawford *et al.* 2014a).

With the loss of suitable nesting habitat, terrapins may nest in suboptimal habitats, or concentrate nests into smaller areas, which can lead to higher nest predation rates (Roosenburg & Place 1995). In particular, vegetation cover may interact with predator abundance to affect terrapin nest success. Burger (1977) reports high mammalian depredation of terrapin nests in wooded shrub and edge habitats, and Hackney (2010) found that terrapin nests in shrub or edge habitats closer to marshes had a higher probability of being depredated compared with nests in open sandy areas farther from the marsh. The interactive effects of vegetation on terrapin hatchling sex ratios and nest predation have the potential to negatively impact terrapin populations, but afford a potential management intervention opportunity to reduce these threats and potentially compensate for other sources of mortality.

The objectives of this study were to measure the effects of vegetation structure and constructed nesting mounds on nest survival of terrapins in the field. Specifically, we evaluated predation rates of simulated nests in areas of managed open grass versus dense shrub (hedgerows) where terrapins nested naturally. We also measured nest predation rates in presumed predator-proof nest boxes on constructed nesting mounds (Buhlmann & Osborn 2011). We hypothesized that the presence of predator guards on nest mounds and the absence of shrub cover would result in higher nest survival of simulated nests. This study was completed as part of a larger experimental study estimating the effects of roadside vegetation on hatchling sex determination in addition to nest success (see Grosse *et al.* 2015).

STUDY SITE

Jekyll Island is a 2,306-ha (5,698-acre) barrier island located in Glynn County, Georgia. Access to Jekyll Island is along an 8.7-km, paved causeway that bisects the marsh. The shoulder of the Downing–Musgrove Causeway (aka Jekyll Island Causeway: JIC) is a raised, dredge-spoil–deposited area, above the high tide line, that provides attractive nesting habitat for female diamondback terrapins. Female terrapins nest along the JIC from April to July, with individuals producing up to two clutches of 4–13 eggs (mean = 7 eggs) each year (Seigel 1980; Zimmerman 1989). A hedgerow of predominantly cedar *Juniperus virginiana* and wax myrtle *Myrica cerifera* occurs along most of the causeway adjacent to the high marsh, and is managed to serve as a windbreak. However, the shrub layer has been removed along some sections of the causeway, creating gaps maintained as short herbaceous (grassy) vegetation up to the high marsh. Between the hedgerow and road there is a parallel strip of maintained (mowed) short grass and other herbaceous plants. As a result, there are two general types of nesting habitat available for diamondback terrapins along the road shoulder: 1) open, regularly mowed, grassy habitat, and 2) shaded hedgerow. In 2009, we created a third habitat type by removing sections of the hedgerow and installing artificial nesting areas (mounds of sandy soil with large predator excluder cages on road shoulders; Figure 2.1; see Buhlmann & Osborn 2011) designed to allow terrapins access to the center of each nesting mound while excluding mammalian and avian predators. Preliminary monitoring demonstrated that female terrapins would nest on nest mounds and inside the caged areas (nest boxes) on the nest mounds.

METHODS

Predation of simulated turtle nests

Using store-bought chicken eggs to simulate turtle nests, we estimated habitat-specific predation rates on the JIC. We placed a nest, consisting of a single chicken egg not protected by caging, in one of the three habitat types on JIC road shoulders. In each habitat type, we placed four nests within a 300-m transect along the roadside. We replicated this design spatially across four 300-m transects and then temporally across four time periods (approx. 2 wk apart) spanning the majority of the terrapin nesting season at our site (sample unit = nest; total $N = 192$). Because we suspected raccoons were entering nest boxes, we placed two simulated chicken egg nests inside each of four nest boxes during the latter two time periods to also estimate predation rates within the nest boxes ($N = 16$). For each nest, we dug a chamber 8–10 cm in depth, buried a single egg, and poured approximately 250 mL of scented water obtained from aquaria containing captive *M. terrapin* over the nest to mimic the release of bladder water by the female at the time of nesting (see Marchand & Litvaitis 2004b; Foley et al. 2012). We spaced nests in the latter three time periods ≥3 m from any previous nest location, and we marked the location of each nest with a handheld Global Positioning System unit and a small flag marker placed 2 m from the nest. Other studies report that up to 98% of depredated terrapin nests were taken within the first 48 h of nesting events (Burger 1977; Goodwin 1994; Butler *et al.* 2004; Munscher *et al.* 2012), so we monitored simulated nests daily for 11 d for evidence of predation. After day 11, we excavated all nondepredated nests to confirm whether chicken eggs were still intact. We wore latex gloves during all phases of the experiment to limit human scent left at simulated nests.

Statistical analyses

We conducted generalized linear models with a logit-link function in R (R Development Core Team 2014) to determine the influence of habitat type and time period on the probability of nest predation. We originally included transect as a random effect in the mixed model, but the variance estimate was zero, which indicated low between-transect variability; therefore, we removed this factor from the model. The unit of analysis was the fate (depredated or not) of an individual nest, and we assumed independence between nests. This assumption was likely met in the open and hedge treatment groups where distances between nests were >50 m. However, nests on mounds were approximately 4 m apart and likely subjected to similar predation risks as adjacent nests. Because nesting habitat on artificial mounds is inherently more confined than natural habitats, we proceeded with the analysis to assess predation risk associated with these management devices. We first tested the full model, including habitat type, time period, habitat × time interaction, and transect as predictor variables and dropped nonsignificant factors ($\alpha = 0.05$) in a backward stepwise procedure using likelihood-ratio tests. We performed post hoc Wald's tests to make pair-wise comparisons between levels within significant factors using the reduced model.

RESULTS

Of 208 simulated chicken egg nests, 157 (75.4%) were depredated within the 11-d monitoring period. The majority (58.0%) of nest predation events occurred within 24 h of nest placement on the JIC, with only 5.7% of predation occurring after 5 d. Mean (± SE) nest predation rates were 45.2% ± 6.3% in open grassy areas, 84.4% ± 5.4% in hedgerows, and 95.3% ± 2.6% on artificial nest mounds. The mean predation rate of nests placed inside nest boxes was 81.3% ± 2.9%. Predation rate was significantly affected by habitat type and time (likelihood-ratio tests [LRT], $\chi^2_2 = 48.08$, $P < 0.001$ and $\chi^2_3 = 31.80$, $P < 0.001$, respectively), while the habitat × time interaction and transect had nonsignificant effects (LRT, $\chi^2_6 < 2.47$, $P > 0.871$) and were dropped from the final model. Nests were significantly more likely to be depredated in hedgerows and on nest mounds relative to open habitats, and nest predation rates significantly increased in later time periods across habitat types (Table 2.1; Figure 2.2).

DISCUSSION

Our results show that habitat affects nest depredation rates; nest predation rates were significantly lower in open grass habitats and higher in hedge and constructed nest mounds. Other studies have reported interactions between vegetation cover effects on terrapin nest predation (Roosenburg & Place 1995), and our results are consistent with other studies that link higher nest depredation (particularly by raccoons) to increased shrub and tree cover around nesting areas (Burger & Montevecchi 1975; Goodwin 1994; Roosenburg 1994; Feinberg and Burke 2003; Ner 2003; Scholz 2006; Hackney 2010; Wnek 2010). Collectively, these studies illustrate an important threat to terrapin populations as well as a potential area of management opportunity.

High predation on nests and hatchlings is a well-documented threat to terrapins and other turtle species (Feinberg & Burke 2003; Butler *et al.* 2006; Munscher *et al.* 2012). Munscher *et al.* (2012) demonstrated that raccoon removal could dramatically improve nest survival. However, high nest predation returned within 1 y of following cessation of raccoon control. Thus, raccoon control can be effective but requires sustained effort. Our results suggest that terrapin nest survival can be moderately high in open grass habitats in areas of high raccoon abundance, and suggest that the removal of shrub and tree vegetation around nesting areas could be an effective long-term management strategy for reducing raccoon predation. Also, removal of shrub cover for raccoons on the narrow causeway may reduce raccoon abundance. From a companion study (Grosse *et al.* 2015), we also estimated a higher percentage of hatchlings were females in experimental nests placed in open areas and on mounds relative to hedgerows. Thus, maintaining or restoring open-grass nesting habitats should increase the production of female hatchlings, which is critical to the growth and recovery of turtle populations (Congdon *et al.* 1994; Mitro 2003; Crawford *et al.* 2014a). We also suggest that creating grass-dominated nesting habitats to increase the production of female hatchlings can be an important tool within an integrated management framework to offset other factors such as roads that cause high adult female mortality. For example, adult female mortality on the JIC ranges from 4 to 16% annually, which is sufficient to cause the population to decline (Crawford *et al.* 2014a). Crawford *et al.* (2014a) estimate that increasing nest success to 50% and the proportion of female hatchlings to 85% could stabilize the population when coupled with modest decreases in road mortality.

The results of our study suggest that maintaining and expanding open-grass nesting habitat could achieve both of these management targets, and in combination with efforts to reduce road mortality, could restore and sustain terrapin populations along the JIC. However, further study is warranted, because of the remaining concern that female terrapins on nesting forays will walk through the narrow grass-dominated areas on the roadsides and access the roadways, which they presumably perceive as higher, well-drained areas. Discouraging terrapin access to the roadway by intercepting them on their nesting foray was the initial intent of the constructed nest mounds. Reducing female terrapin road mortality must still remain a priority to prevent further population declines.

We posit that higher predation rates of simulated nests within the hedgerows and on nest mounds occurred for two reasons. First, nest mounds were placed in areas of high historical nesting densities (T. Norton, personal communication) and current hot spots of nesting activity (Crawford *et al.* 2014b), which is consistent with other studies that report higher predation rates in habitats with higher nest densities (e.g., Roosenburg & Place 1995; Feinberg & Burke 2003; Marchand & Litvaitis 2004a). Terrapins show high nest fidelity between years (Goodwin 1994; Sheridan 2010; Crawford *et al.* 2014b), and predators may learn to target these areas. Secondly, hedgerows form extended corridors for predator movement, particularly along roadways, and constructed nest mounds in the study were initially placed in small gaps along the hedgerows. The nest boxes placed on top of the artificial nest mounds were intended to provide nest protection from predators. However, raccoons at our study site demonstrated the ability to squeeze through the box opening and depredate terrapin nests. We followed the nest box design of Buhlmann and Osborn (2011), which was successful at improving nest survival for other turtle species. Nonetheless, the effectiveness of nest boxes on artificial nest mounds remains unclear, but the nest boxes have undergone structural modification to ensure they are effectively excluding potential predators. If modified successfully, artificial nesting habitat with predator exclusion structures could be an effective tool for improving nest success at hotspots of terrapin nesting and road mortality (see Crawford *et al.* 2014b).

We caution that our advocacy for increasing open, grassy habitat must consider other factors that could degrade nesting habitat quality for terrapins. Management activities to remove shrubs or trees must not substantially change the soil composition or increase the potential for erosion within the nesting area. Terrapins nest in sandy soils with large particle size that improves gas diffusion and has a lower water potential, which reduces hydric constraints on developing embryos (Roosenburg 1994; Wnek 2010). Activities that increase organic content of soils or compact the soils would likely reduce terrapin egg survival (Wnek 2010). In addition, to avoid killing or disturbing nesting females, mowing to maintain open, grassy habitats should not occur during the nesting season. Care should also be taken to manage the types of herbaceous plants that replace shrubs and trees in restored habitats. The dense planting of either grasses to control erosion or of invasive plants are both known to increase terrapin nest failure (Roosenburg 1991; Wnek 2010). Some plant species have roots that can infiltrate and kill terrapin eggs (Lazell & Auger 1981), and dense grasses can reduce soil moisture potential, resulting in higher egg-failure rates. The creation of open-grass–dominated areas suitable for nesting should include clump grasses with open sandy areas in between.

In conclusion, we demonstrated that vegetation management around terrapin nesting habitat interacts to affect nest predation rates, as well as hatchling sex ratios (Grosse *et al.* 2015). Both the intentional planting of shrubs and trees as hedgerows or windbreaks along causeways and the unintentional succession of woody plants are common scenarios in developed coastal areas, and may be contributing to terrapin population declines by creating male-biased hatchling sex ratios and facilitating subsidized predators. If conducted properly and in concert with methods to reduce road mortality, the maintenance and restoration of open grassy nesting areas can be a highly feasible and effective component of an integrated management plan to restore and sustain terrapin populations.

ACKNOWLEDGMENTS

We thank Linda Pirie, Lisa Rodriguez, Shaylin Duncan, Simon Diltz, Steven Nelson, Amy Hupp, Rachel Thomas, and the Georgia Sea Turtle Center staff for their help in every aspect of this project. This research is partially supported by AGL Resources Foundation through the Jekyll Island Foundation, which raises funds for projects related to the conservation, preservation and education of Jekyll Island.

REFERENCES

Buhlmann KA, Osborn CP. 2011. Use of an artificial nesting mound by wood turtles (*Glyptemys insculpta*): A tool for turtle conservation. *Northeastern Naturalist* 18:315–334.

Burger J. 1976. Temperature relationships in nests of the northern diamondback terrapin, *Malaclemys terrapin terrapin*. *Herpetologica* 32:412–418.

Burger J. 1977. Determinants of hatching success in diamondback terrapin (*Malaclemys terrapin*). *American Midland Naturalist* 97:444–464.

Burger J, Montevecchi WA. 1975. Nest site selection in the terrapin *Malaclemys terrapin*. *Copeia* 1975:113–119.

Butler JA, Broadhurst C, Green M, Mullin Z. 2004. Nesting, nest predation and hatchling emergence of the Carolina diamondback terrapin, *Malaclemys terrapin centrata*, in northeastern Florida. *American Midland Naturalist* 152:145–155.

Butler JA, Seigel RA, Mealey BK. 2006. *Malaclemys terrapin*—diamondback terrapin. Pages 279–295 in Meylan P, editor. Biology and conservation of Florida turtles. *Chelonian Research Monographs* 3.

Chambers RM, Maerz JC. *In review*. Terrapin bycatch in the blue crab fishery. In Roosenburg WM, Kennedy VS, editors. Ecology and conservation of the diamondback terrapin. Baltimore, Maryland: Johns Hopkins University Press.

Congdon JD, Dunham AE, van Loben Sels, RC. 1994. Demographics of common snapping turtles (*Chelydra serpentina*): Implications for conservation and management of long-lived organisms. *American Journal of Zoology* 34:397–408.

Crawford BA, Maerz JC, Nibbelink NP, Buhlmann KA, Norton TM. 2014a. Estimating the consequences of multiple threats and management strategies for semi-aquatic turtles. *Journal of Applied Ecology* 51:359–366.

Crawford BA, Maerz JC, Nibbelink NP, Buhlmann KA, Norton TM, Albeke SE. 2014b. Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology* 51:367–375.

Cunningham B, Woodward MW, Pridgen J. 1939. Further studies of turtle (M*alaclemys centrata Lat.*) eggs. *American Naturalist* 73:285–288.

Dorcas ME, Willson JD, Gibbons JW. 2007. Crab trapping causes population decline and demographic changes in diamondback terrapins over two decades. *Biological Conservation* 137:334–340.

Feinberg JA, Burke RL. 2003. Nesting ecology and predation of diamondback terrapins, *Malaclemys terrapin*, at Gateway National Recreation Area, New York. *Journal of Herpetology* 37:517–526.

Foley SM, Price SJ, Dorcas ME. 2012. Nest-site selection and nest depredation of semi-aquatic turtles on golf courses. *Urban Ecosystems* 15:489–497.

Gibbons JW, Greene JL. 1979. X-ray photography: a technique to determine reproductive patterns of freshwater turtles. *Herpetologica* 35:86–89.

Gibbons JW, Lovich JE, Tucker AD, Fitzsimmons NN, Greene JL. 2001. Demographic and ecological factors affecting conservation and management of the diamondback terrapin (*Malaclemys terrapin*) in South Carolina. *Chelonian Conservation and Biology* 4:66–74.

Goodwin CC. 1994. Aspects of nesting ecology of the diamondback terrapin (*Malaclemys terrapin*) in Rhode Island. Master's thesis. Kingston: University of Rhode Island.

Grosse AM, Maerz JC, Hepinstall-Cymerman J, Dorcas ME. 2011. Effects of roads and crabbing pressures on diamondback terrapin populations in coastal Georgia. *Journal of Wildlife Management* 75:762–770.

Hackney AD. 2010. Conservation biology of the diamondback terrapin in North America: Policy status, nest predation, and managing island populations. Master's thesis. Columbia, South Carolina: Clemson University.

Hart KM, Crowder LB. 2011. Mitigating by-catch of diamondback terrapins in crab pots. *Journal of Wildlife Management* 75:264–272.

Herlands R, Wood R, Pritchard J, Clapp H, Le Furge N. 2004. Diamondback terrapin, (*Malaclemys terrapin*) head-starting project in southern New Jersey. Pages 13–18 in Swarth C, Roosenburg WM, Kiviat E, editors. Conservation and ecology of turtles of the Mid-Atlantic region. Salt Lake City, Utah: Bibliomania.

Hernandez-Divers SJ, Stahl SJ, Farrell R. 2009. An endoscopic method for identifying sex of hatchling Chinese box turtles and comparison of general versus local anesthesia for coelioscopy. *Journal of the American Veterinary Medical Association* 234:800–804.

Jeyasuria P, Place AR. 1997. Temperature-dependent aromatase expression in developing diamondback terrapin (*Malaclemys terrapin*) embryos. *Journal of Steroid Biochemistry and Molecular Biology* 61:415–425.

Jeyasuria P, Roosenburg WM, Place AR. 1994. Role of P450 aromatase in sex determination of the diamondback terrapin, *Malaclemys terrapin*. *Journal of Experimental Zoology* 270:95–111.

Lazell JD, Auger PJ. 1981. Predation on diamondback terrapin (*Malaclemys terrapin*) eggs by dunegrass (*Ammophila breviligulata*). *Copeia* 1981:723–724.

Mader DR, editor. 2006. Reptile medicine and surgery. 2nd edition. St. Louis: Elsevier.

Maerz, JC, Seigel RA, Crawford BA. *In review*. Terrapin conservation: mitigating habitat loss, road mortality, and subsidized predators. In Roosenburg WM, Kennedy VS, editors. Ecology and conservation of the diamondback terrapin. Baltimore, Maryland: Johns Hopkins University Press.

Marchand MN, Litvaitis JA. 2004a. Effects of habitat features and landscape composition on the population structure of a common aquatic turtle in a region undergoing rapid development. *Conservation Biology* 18:758–767.

Marchand MN, Litvaitis JA. 2004b. Effects of landscape composition, habitat features, and nest distribution on predation rates of simulated turtle nests. *Biological Conservation* 117:243–251.

Mitro MG. 2003. Demography and viability analysis of a diamondback terrapin population. *Canadian Journal of Zoology* 81:716–726.

Munscher EC, Kuhns EH, Cox CC, Butler JA. 2012. Decreased nest mortality for the Carolina diamondback terrapin (*Malaclemys terrapin centrata*) following the removal of raccoons (*Procyon lotor*) from a nesting beach in northeastern Florida. *Herpetological Conservation and Biology* 7:167–184.

Ner SE. 2003. Distribution and predation of diamondback terrapin nests at six upland islands of Jamaica Bay and Sandy Hook Unit, Gateway National Recreation Area. Master's thesis. Hempstead, New York: Hofstra University.

R Development Core Team. 2014. A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing. http://www.R-project.org (January 2014).

Roosenburg WM. 1991. The diamondback terrapin: habitat requirements, population dynamics, and opportunities for conservation. Pages 234–237 in Chaney A, Mihursky J, editors. New perspectives in the Chesapeake system: a research and management partnership. Solomons, Maryland: Chesapeake Research Consortium Publication Number 137.

Roosenburg WM. 1994. Nesting habitat requirements of the diamondback terrapin: A geographic comparison. *Wetlands Journal* 6:9–12.

Roosenburg WM, Place AR. 1995. Nest predation and hatchling sex ratio in the diamondback terrapin: implications for management and conservation. Pages 65–70 in S Nelson, editor. Toward a sustainable coastal watershed: the Chesapeake experiment. Solomons, Maryland: Chesapeake Research Consortium Publication Number 149.

Scholz AL. 2006. Impacts of nest site choice and nest characteristics on hatching success in the diamondback terrapins of Jamaica Bay, New York. Master's thesis. Hempstead, New York: Hofstra University.

Seigel RA. 1980. Nesting habits of diamondback terrapins (*Malaclemys terrapin*) on the Atlantic coast of Florida. *Transactions of the Kansas Academy of Sciences* 83:239–246.

Seigel RA. 1993. Apparent long-term decline in diamondback terrapin populations at the Kennedy Space Center, Florida. *Herpetological Review* 24:102–103.

Sheridan CM, Spotila JR, Bien WF, Avery HW. 2010. Sex-biased dispersal and natal philopatry in the diamondback terrapin, *Malaclemys terrapin*. *Molecular Ecology* 19:5497–5510.

Szerlag S, McRobert SP. 2006. Road occurrence and mortality of the northern diamondback terrapin. *Applied Herpetology* 3:27–37.

Winters J. 2013. The effects of bulkheading on diamondback terrapin nesting in Barnegat Bay, New Jersey. Doctoral dissertation. Philadelphia, Pennsylvania: Drexel University.

Wnek JP. 2010. Anthropogenic impacts on the reproductive ecology of the diamondback terrapin, *Malaclemys terrapin*. Doctoral dissertation. Philadelphia, Pennsylvania: Drexel University.

Wood R, Herlands R. 1997. Turtles and tires: The impact of roadkills on northern diamondback terrapin, *Malaclemys terrapin terrapin*, populations on the Cape May Peninsula, southern New Jersey, USA. Pages 46–53 in Abbema JV, editor. Conservation, restoration, and management of tortoises and turtles—an international conference. New York: New York Turtle and Tortoise Society.

Zimmerman TD. 1989. Latitudinal reproduction variation of the salt marsh turtle, the diamondback terrapin (*Malaclemys terrapin*). Master's thesis. Charleston, South Carolina: The College of Charleston.

Table 2.1. Generalized linear model (logit-link, binomial distribution) estimates and odds ratios for habitat type and time period effects on predation rates of simulated turtle nests on the Jekyll Island Causeway, Georgia, USA.

				Odds Ratios		
					95% Confidence	
Variable	β	SE	*z*	Estimate[a]	Intervals	
Intercept (Open, Time 1)	-3.063	0.682	-4.49*	0.17^{A}	0.06	0.43
Hedgerows	2.438	0.522	4.67*	11.33^{B}	4.33	33.65
Mounds	3.898	0.723	5.39*	47.59^{B}	13.42	237.53
Time 2	0.733	0.551	1.33	2.08^{AB}	0.72	6.36
Time 3	2.093	0.630	3.32*	8.11^{ABC}	2.50	30.12
Time 4	3.399	0.784	4.33*	29.94^{C}	7.29	166.67

* $P < 0.001$

[a] Letters indicate significant differences ($p < 0.05$) in predation rates between levels within the same factor (e.g., Open and Hedgerows, Time 1 and 4).

Figure 2.1. View of the Jekyll Island Causeway and 3 nesting habitat types: hedgerow and open habitats (above), and nesting mound with a nest box (below).

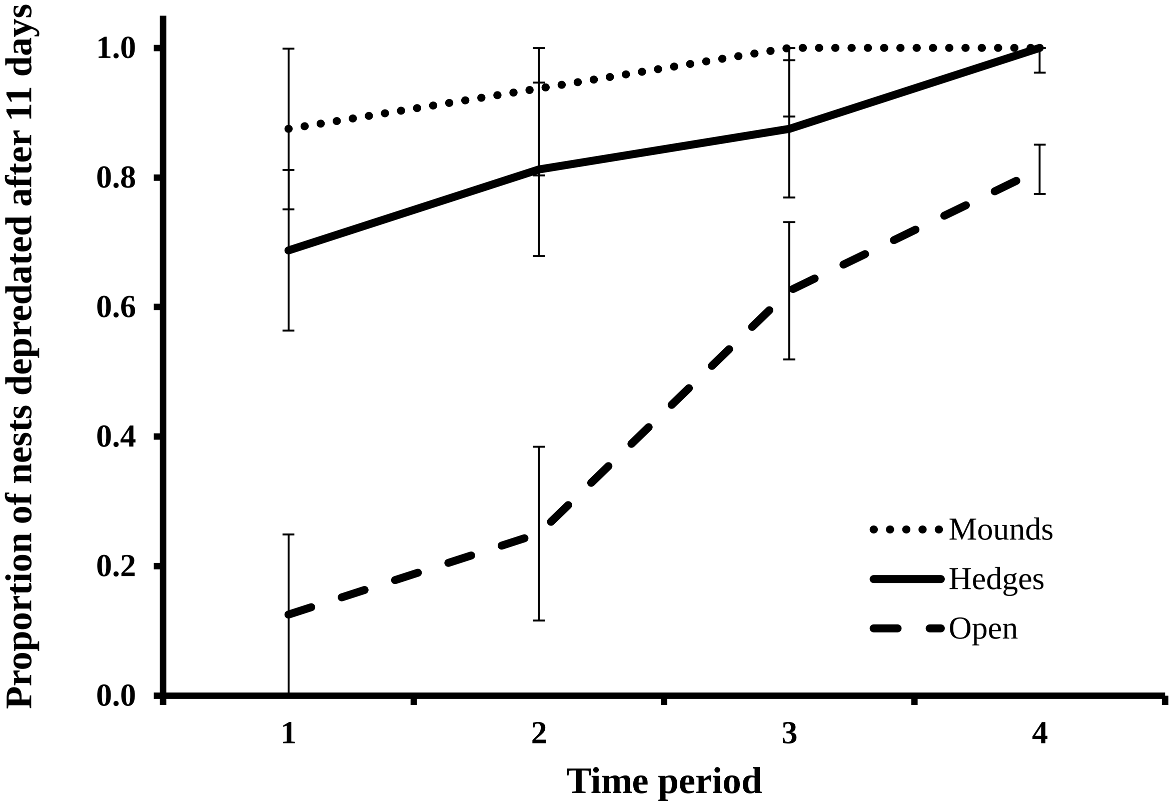


Figure 2.2. Mean predation rates (± SE) of plots with chicken egg (simulated turtle) nests (N = 48) in three habitats on the Jekyll Island Causeway throughout May to July 2011.

CHAPTER 3

WHEN DRIVERS AND TERRAPINS COLLIDE: ASSESSING STAKEHOLDER ATTITUDES TOWARD WILDLIFE MANAGEMENT ON THE JEKYLL ISLAND CAUSEWAY[1]

---

[1] Crawford, B.A., N.C. Poudyal, and J.C. Maerz. 2015. Human Dimensions of Wildlife: An International Journal. 20:1-14.
This is an Accepted Manuscript of an article published in Human Dimensions of Wildlife online [January 7, 2015], available online: http://tandfonline.com/doi/full/10.1080/10871209.2014.948103
Reprinted here with permission of the publisher.

ABSTRACT

Roads are a pervasive fixture on most landscapes and frequently impact wildlife. Management of road effects on wildlife should consider the attitudes and areas of contention among various stakeholders using the road. To inform future management of road impacts on the diamondback terrapin (*Malaclemys terrapin*), a species of conservation concern, we surveyed stakeholder groups on Jekyll Island, Georgia, USA to measure overall attitudes and value orientations relating to terrapin management and to evaluate the acceptability and potential for conflict among user groups regarding specific management alternatives. A majority of respondents supported the protection of terrapins, though support varied by group membership and value orientation. Specific actions that did not impact speed limits or road aesthetics were most acceptable, but substantial conflict existed for most strategies. We intend to use these results to improve communication and buy-in of stakeholders as we develop management alternatives for the diamondback terrapin on Jekyll Island.

INTRODUCTION

Natural resource decisions promoting biodiversity are inherently difficult because they must consider complex social, political, and economic perspectives from multiple groups with a stake in the decision (McShane et al., 2011). Incorporating stakeholders' perspectives has become a prominent, if not normative, feature of wildlife management (Riley et al., 2002). In order to produce long-lasting conservation strategies, managers should integrate ecological and human dimensions knowledge that can help identify and reduce areas of conflict, develop creative solutions, and facilitate negotiations among multiple stakeholder groups (McShane et al., 2011).

To predict public support for conservation decisions, human dimensions researchers have increasingly used the cognitive hierarchy, a conceptual framework that includes values, value orientations, attitudes, and behaviors (Fulton, Manfredo, & Lipscomb, 1996; Vaske & Donnelly, 1999; Zinn, Manfredo, Vaske, & Wittman, 1998). Individuals' wildlife value orientations have been identified as strong predictors of attitudes and reported behaviors toward specific conservation actions (see Fulton et al., 1996; Steel, List, & Shindler, 1994; Vaske & Donnelly, 1999). Value orientations are relatively stable and exist along a continuum from anthropocentric (i.e., a utilitarian, human-centered view of natural resources) to biocentric (i.e., a view that nature has inherent value in equal standing with human-centered values; Steel et al., 1994; Thompson & Barton, 1994; Vaske & Donnelly, 1999). Alternatively, attitudes are more dynamic and represent one's feelings of favor or disfavor toward a specific object or idea (e.g., Thompson & Barton, 1994). Previous studies have repeatedly found that attitudes can directly predict as well as mediate the influence of value orientations on behaviors in natural resource contexts, such as determining hunting and fishing preferences (Fulton et al., 1996) or voting intentions (Vaske &

Donnelly, 1999). Few studies, however, have assessed the public's attitudes and underlying value orientations in the context of managing a growing threat to wildlife: roads.

Roads are a public asset important to people's daily lives and to local, regional, and national economic growth. In the conterminous United States, 82% of all land area is within 1 km of the nearest roadway (Riitters & Wickham, 2003). Roads, however, present a major management challenge to the conservation of wildlife through negative impacts, including the reduction and fragmentation of wildlife habitat, predation by species subsidized by human activities (e.g., raccoons; *Procyon lotor*), impediment to animal movement, and alteration of species behavior (Fahrig & Rytwinski, 2009; Trombulak & Frissell, 2000). The most direct and ubiquitous road-associated threat to wildlife, however, is collisions with vehicles (hereafter referred to as road mortality; Fahrig & Rytwinski, 2009). Although declining species in landscapes fragmented by roads often face several threats simultaneously, failing to address negative impacts directly caused by road mortality may compromise the ability of any management practice to facilitate recovery.

Strategies to manage road mortality on wildlife are often scrutinized by public users or stymied by state or federal regulations. Most management actions (e.g., altering speed limits, constructing barriers or culverts, or restricting vehicle access) have direct or unforeseen consequences to drivers' behaviors, happiness, or safety. For example, drivers may have unfavorable attitudes toward strategies that increase travel time, reduce driver safety, or infringe on other amenities such as scenic views. Moreover, management decisions must account for diverging preferences among multiple, distinct groups of road users. The public may view management unfavorably if they do not know of or believe evidence supporting conservation actions (Messmer, Brunson, Reiter, & Hewitt, 1999; Shindler, Brunson, & Cheek, 2004), or stakeholders may find the consequences, costs, or duration of action unacceptable (Stankey & Shindler, 2006; Stout, Knuth, & Curtis, 1997). Formally measuring road users' attitudes is necessary to reveal potential conflicts and inform the selection of strategies that are both biologically effective and socially acceptable.

Most studies of drivers' preferences for road management have focused on vehicle collisions with large ungulates and have operated with the impetus to reduce risks to human safety or property damage (Marcoux, 2005; Stout et al., 1997; Sullivan & Messmer, 2003; Whittaker et al., 2001). In contrast, social acceptability of management strategies targeting threatened or charismatic species has predominately been assessed for issues separate from roads including management of private lands (Jacobson & Marynowski, 1997), removal of predators (Karlsson & Sjöström, 2007; Messmer et al., 1999) or invasive alien species (Sharp, Larson, & Green, 2011), or reduction of boat-wildlife collisions (Aipanjiguly, Jacobson, & Flamm, 2003). To our knowledge, only one human dimensions study (Vaske & Donnelly, 2007) has examined vehicle collisions involving small-bodied vertebrates – where the threat posed minimal risk to humans but was expected to cause wildlife population declines in the absence of management efforts. However, we note that the objective of this study was to inform education programs rather than to guide management alternatives.

The diamondback terrapin (*Malaclemys terrapin*) is a small, estuarine turtle that inhabits salt marshes along the Eastern and Gulf Coasts of the United States – regions experiencing the fastest annual increases in road densities and traffic loads from tourists, residents, and employees (Baird, 2009). Subsequently, terrapin populations are declining due to multiple anthropogenic threats including road mortality (Maerz, Seigel, & Crawford, *In review*). By fragmenting salt marsh habitat, roadsides provide open, elevated nesting sites attractive to adult females, which are sometimes struck by vehicles while attempting to cross roads each summer from April through July (Maerz, Seigel, & Crawford, *In review*). In addition to mortality from vehicles, predators such as raccoons (*Procyon lotor*) can cause high rates (50%-90%) of mortality for eggs laid on road shoulders (Feinberg & Burke, 2003; Szerlag-Egger & McRobert, 2007). Likely representative of conditions on many heavily-traveled coastal roads throughout the terrapin's range, road mortality and nest predation on the Jekyll Island Causeway (JIC) leading to Jekyll Island, GA, USA were both contributing to terrapin population declines (Crawford, Maerz, Nibbelink, Buhlmann, & Norton, 2014a). Because terrapin populations are expected to decline as pressure from development and tourism increases in coastal areas, management solutions are needed that recover and stabilize terrapins, as well as other wildlife affected by roads.

The general objective of this study was to understand attitudes and acceptability of road and terrapin management for patron groups using the JIC. Specific objectives were to: (a) measure the public's general attitudes toward management and their wildlife value orientations, (b) understand factors influencing attitudes, (c) rank proposed management alternatives by public acceptability, and (d) identify conflicts among groups toward any action. We hypothesized that JIC users with relatively biocentric value orientations or first-hand experience with terrapin mortality would be more supportive of terrapin management than their respective counterparts. We also hypothesized that the specific management actions with more severe impacts to driving experience and mesomammal predators would have relatively lower public acceptance, which would relate to higher degrees of conflict among stakeholders.

METHODS

Study Area and Patron Groups

The 8.7-km JIC is a state highway with average annual daily traffic (AADT) of 3,440 vehicles/day (Georgia Department of Transportation, 2011), and it is the only road providing access to the island. Prior to data collection, we targeted visitors, residents, and employees of Jekyll Island as distinct patron strata that use the JIC and are stakeholders in future management decisions. An estimated 1.25 million annual visitors travel to Jekyll Island, and monthly JIC traffic volumes peak from May through July corresponding to increased summer tourism (E. Garvey, Jekyll Island Authority [JIA]; unpubl. data). There are an estimated 805 residents of Jekyll Island (U.S. Census Bureau, 2010). Lastly, 120 people are employed by JIA – the agency commissioned by the Georgia state government with overseeing the development of Jekyll Island while protecting its natural resources. JIA employees include elected officials, Georgia Sea Turtle Center (GSTC) staff, and workers at a network of businesses including hotels, restaurants, golf courses, and attractions.

Survey Methodology

To measure stakeholders' attitudes toward management and factors influencing these attitudes, we surveyed island visitors, residents, and employees with a variety of sampling methods in 2012. For visitors, we sampled every other person (over the age of 18) intercepted at popular recreation hotspots on Jekyll Island (beaches, a fishing pier, and picnic areas) using in-person, self-administered surveys randomly stratified across sites, starting times (1000 and 1500 hrs), and weekdays (Monday through Thursday) and weekends (Friday through Sunday). Participants were generally stationary at sites (e.g., sitting on a beach) and took 5-10 minutes to complete the survey, which yielded a high response rate of 97.2%. For residents and employees, we followed Dillman's (2007) methods of construction to compose an online version (using SurveyMonkey™) of the same survey used for visitors, and we recruited respondents via listserves containing 590 resident members of the Jekyll Island Citizens Association (JICA: http://www.jekyllcitizens.org/) and 120 employee members of JIA (http://www.jekyllisland.com/jekyll-island-authority/). Response rates were relatively low for residents (23.3%) and employees (47.5%), so we recruited additional participants by intercepting them at randomly selected residences and businesses between 13 and 15 December 2012.

The survey questionnaire included items developed from existing literature (e.g., Vaske & Donnelly, 2007). We measured the following factors for this study: (a) attitudes toward general management, (b) wildlife value orientations, (c) acceptability of specific management alternatives, (d) past experience on Jekyll Island, including experience with terrapin-vehicle collisions and participation in wildlife-related activities, and (e) sociodemographic characteristics. Using 5-point scales (2 – Strongly Agree, -2 – Strongly Disagree), we measured respondents' attitudes toward general management statements (e.g., We should protect terrapins from being struck by vehicles) and value orientation statements (e.g., Terrapins have as much right as people to exist, Management should focus on doing what is best for island patrons over what is best for terrapins) developed from previous studies (Fulton et al., 1996; Vaske & Donnelly, 1999). We used a similar scale of acceptability (2 – Highly Acceptable, -2 – Not at all Acceptable) to measure attitudes toward specific management actions (Table 3.1). Other questions regarding patrons' knowledge and experience with terrapins used a dichotomous checked/un-checked response format. The complete questionnaire is presented in Appendix A.

Data Analysis

We conducted the reliability analysis and discriminant validity test of two multi-item scales designed to measure attitudes and value orientations in SPSS (SPSS Inc., 2007) using Cronbach's alpha and Principle Axis Factoring (PAF). We used Bartlett's Test of Sphericity and the Kaiser-Meyer-Olkin (KMO) Measure of Sampling Adequacy to confirm that PAF was appropriate given the survey data. Once we determined scale reliability, we calculated composite scores for each respondent for general attitudes toward terrapin management and value orientations (based on 3 and 7 items, respectively). We calculated mean scores for general attitudes and value orientations across all respondents using pairwise deletion for omitted items. We used one-way ANOVAs to evaluate attitudes toward general management and wildlife value orientations across patron groups followed by post-hoc Tukey's HSD tests for pairwise comparisons (Sokal & Rohlf, 1995). Next, we performed logistic regression to analyze people's general attitudes toward terrapin management as a function of value orientations, experience relating to Jekyll Island and terrapin mortality, and sociodemographic factors, including group membership, using listwise deletion.

Lastly, we calculated mean acceptability ratings across all respondents to rank the 11 specific management actions and used one-way ANOVAs followed by Tukey's HSD tests to evaluate different ratings between groups. We also evaluated conflict over general terrapin management and specific actions using the second generation Potential for Conflict Index (PCI_2), developed by Vaske, Beaman, Barreto, and Shelby (2010), which displays the central tendency and dispersion of a given group's attitudes. PCI_2 ranges from 0 (maximum consensus in management acceptability) to 1 (maximum conflict) for each group's attitudes. PCI_2 values provide useful measures of the relative degree of conflict within a group's respondents, but methods for interpreting these metrics based on standardized critical values have not been established to date (Vaske et al., 2010). We also calculated statistical differences (*d*) in PCI_2 values between groups using tests available in PCI_2 software (described in Vaske et al., 2010), and *d* statistics were compared to a Bonferonni-corrected critical value (2.12) for a normal distribution to assess significant differences among 3 groups' attitudes at $\alpha = .017$.

RESULTS

Response Rates and Respondents

Out of 1,893 patrons contacted/approached, we received responses from 1,043 visitors, 214 residents, and 87 employees of Jekyll Island (Total $N = 1{,}344$; overall response rate = 71%). The majority of respondents tended to be return visitors to the island, had visited JI nature centers including the GSTC, and had not seen a terrapin struck by a vehicle. Demographically, surveyed patrons tended to be 40 years of age or older, female, Caucasian, and were higher educated on average (Table 3.2).

Attitude and Value Orientation Scales

KMO measures of sampling adequacy showed that both general attitudes and value orientation scales were appropriate for factor analysis (KMO = .763 and .840, respectively). PAF analysis of the multi-item scale for general attitudes toward terrapin management supported a one-factor solution and acceptable level of internal consistency (Cronbach's alpha = .717) based on criteria of Tavakol and Dennick (2011). All three items of support of general management were retained (smallest factor loading = .504) and, together, explained 64% of the variance. We averaged these items into composite scores (from -2 to 2) and created a binary variable for logistic regression analysis representing if a patron supported (rounded average score = 1 or 2) or did not support (rounded average score = -2, -1, or 0) management (see Poudyal, Siry, &

Bowker, 2010). PAF analysis of the value orientation scale supported a two-factor solution. Of nine items, seven were included in the first factor (smallest factor loading = .407) and explained 46% of the variance. The two items included in the second factor (We should maintain views of the marshland from the Causeway regardless of impacts on terrapins, We should maintain the current speed limit regardless of the impact on terrapins) only explained 16% of the variance. These items addressed fairly distinct constructs relative to the remaining seven items that focused on general value orientations. Therefore, we only retained and averaged responses for these seven items into a composite score for value orientation (from -2 to 2) representing anthropocentric and biocentric views, respectively. Averaging was supported by high internal consistency among these seven items (Cronbach's alpha = .830). Following the procedures used for general attitudes, we converted this score to a binary predictor of value orientations in the logistic model.

General Attitudes and Value Orientations (Objective a)

From pooling responses across all stakeholder groups, respondents demonstrated supportive general attitudes to terrapin management (mean composite score = 1.39), with 77% of respondents classified as supportive on the binary scale. Respondents' value orientations were relatively biocentric (mean composite score = 1.11) with 64.3% classified as biocentric on the binary scale, which reflected the belief that terrapins had the right to exist and had some degree of ecological importance. Responses differed among patron groups for general attitudes ($F_{2,\,1224} = 11.05$, $p < .001$) and value orientations ($F_{2,\,1224} = 16.54$, $p < .001$), with employees reporting lower scores for both variables relative to visitors and residents. Still, each group was generally supportive of management (mean scores ≥ 1.02) and reported somewhat biocentric value orientations (mean scores ≥ 0.75).

Predictors of Support (Objective b)

Results from the logistic regression model showed relative influences of predictor variables on patron support for terrapin management, and the model fit the data adequately (Table 3.2). Multicollinearity among predictors was not an issue (Tolerance ≥ 0.89, Variance Inflation Factor ≤ 1.24). The model's classification accuracy rate (77.8%) exceeded the proportional by chance accuracy rate (46.9%). Parameter estimates (β) and odds ratios revealed that value orientation was the strongest predictor of attitudes to terrapin management, and patrons with biocentric, relative to anthropocentric, value orientations were 6.4 times more likely to support management. Visitors and residents were 1.8 and 2.2 times, respectively, more likely to support terrapin management compared to island employees. Visiting JI nature centers positively influenced attitudes toward management while a person's first-hand experience with terrapin mortality on the JIC did not have a significant effect on attitudes ($p = .429$). Sociodemographic factors had variable effects on management attitudes, but patrons that were under 40 years of age, were female, and had received higher education were more likely than their respective counterparts to show support for management.

Management Alternatives: Acceptability and Potential for Conflict (Objectives c and d)

Across respondents, average acceptability of specific management actions ranged from 1.13 (hidden nest boxes) to -1.35 (do nothing) on the -2 to 2 scale (Table 3.1). Mean acceptability scores showed moderate differences among groups for using nest boxes in plain sight, clear-cutting roadside hedges, or relocating raccoons ($F_{2,1302\text{-}1327} = 2.54\text{-}2.87$, $.057 < p < .079$) and significant differences for all other actions ($F_{2,1302\text{-}1327} = 3.37\text{-}73.78$, $p < .017$).

PCI_2 values showed that general terrapin management had the highest acceptability with low conflict among visitors, residents, and employees (Figure 3.1), but specific management options reflected varying degrees of conflict across patron groups. For visitors, relatively low levels of conflict ($PCI_2 \leq .19$) were observed for four acceptable options (using hidden or unhidden nest boxes, short fences, and flashing signage) and one unacceptable option (doing nothing). The remaining options showed higher degrees of conflict (PCI_2 range = .29 to .46). For residents and employees, only two options (hidden or unhidden nest boxes) were rated as acceptable with low levels of conflict ($PCI_2 \leq .24$), while other acceptable options (flashing lights, short fences) reflected higher degrees of discord ($PCI_2 \geq .34$). Relocating or lethally removing raccoons ranked as the two most controversial options for residents and employees ($PCI_2 \geq .49$) and among the most controversial options for visitors ($PCI_2 \geq .35$). PCI_2 values were significantly lower (stronger agreement) for visitors than residents and employees for 6 of 11 management actions. PCI_2 values never significantly differed between residents and employees for any management option. We present results and trends with a reduced list of management options graphically (Figure 3.1).

DISCUSSION

While there has been a growing awareness that road mortality is contributing to declines of species across multiple wildlife taxa and that mitigation strategies are needed pre- and post-road construction (Fahrig & Rytwinski, 2009; Trombulak & Frissell, 2000), management agencies have yet to measure and incorporate public values and attitudes into conservation decision making in most contexts. This is the first study, to our knowledge, that measured public attitudes toward road management in order to design conservation plans for at-risk species other than large ungulates. Our results demonstrate the majority of JIC users supported the protection of terrapins and held relatively biocentric value orientations; however, this support varied by group membership and value orientation. Across all respondents, support varied for specific management alternatives, but actions that did not impact speed limits or road aesthetics were most acceptable. Finally, we found substantial conflict within and among user groups regarding the acceptability of most strategies, especially the lethal removal of raccoons.

The strong support for general management observed in our study agrees with other studies focused on reducing vehicle collisions with large ungulates that pose direct risks to driver safety and vehicle damage (Marcoux, 2005; Stout et al., 1997; Sullivan & Messmer, 2003). However, we emphasize that our study was conducted within a relatively different social context. The presence of small-bodied animals on high-traffic roads may still pose risks to drivers if they swerve to miss (or strike) wildlife or walk on the road to remove an animal. However, we suspect that safety concerns were minimal and did not contribute to patron support since only 22% of respondents had witnessed a terrapin on the causeway, and this factor did not significantly influence patrons' general attitudes. This finding was counter to our hypothesis as well as previous studies that found strong relationships between a person's firsthand experience with a wildlife problem and their acceptability of management (e.g., crop damage and lethal removal of pests; Koval & Mertig, 2004). Alternatively, the majority of patrons had visited local nature centers and held biocentric value orientations, and, confirmatory to our hypotheses and previous studies, these were significant predictors of management attitudes (Fulton et al., 1996; Steel et al., 1994; Vaske & Donnelly, 1999). Thus, patrons were motivated to support management due to values inherent of wildlife – and not just those values linked to personal safety and wellbeing. These results likely reflect Jekyll Island's status as a state park that, compared to more developed coastal destinations, attracts a greater proportion of people with pro-wildlife values and attitudes. Because state parks, national seashores (e.g., Cumberland Island, GA, Assateague Island, MD), and nature centers are common in coastal areas throughout the terrapin’s range, the attitudes of road users in our study may be representative of other sites. Still, management decisions for additional roads impacting terrapins should measure local attitudes, values, and other predictors and investigate how these vary geographically.

While attitudes toward general management indicate support of stakeholder groups, we found less and more variable support for any specific management action, which has strong implications for future decisions. These results agree with several previous studies that found more precise and predictable relationships between specific, as opposed to general, attitudes and targeted conservation-oriented behaviors (McKenzie-Mohr, Nemiroff, Beers, & Desmarais, 1995; St John, Edwards-Jones, & Jones, 2011). Moreover, lethal removal of raccoons was deemed unacceptable by respondents, which supports previous findings that the public is generally opposed to lethal measures for managing wildlife in suburban or urban contexts (e.g., Stout et al., 1997; Zinn et al., 1998). Our results demonstrate that management decisions based solely on the public’s general attitudes to species or threats may still lead to dissatisfied stakeholder groups that first appeared supportive. Assessing attitudes to specific management options prior to implementation will provide decision makers with more accurate and transparent information on which to determine optimal strategies that address biological and social goals (McShane et al., 2011).

When comparing preferences within and across stakeholder groups, substantial contention emerged. Overall, visitors and residents were twice as likely to support general terrapin management compared to employees, but responses from residents and employees showed similar trends in acceptability and contention toward most specific actions. Residents and employees, relative to visitors, were dissatisfied with many proposed actions, and contention within these groups existed even for acceptable actions (e.g., flashing signage). Conflict is expected when management decisions promote the values of certain parties but not others. Anthropogenic changes to the landscape can disenfranchise local residents and employees, who likely differ from visitors in which social or natural amenities are valued (Boucquey et al., 2012). For example, residents and employees travel on the JIC more regularly while about 20% of visitors surveyed have only been to Jekyll Island once. Likewise, management actions listed in our survey that would more permanently alter drivers' commutes (reducing speed limits, constructing speed bumps, or concrete barriers) were significantly less acceptable to residents and employees compared to visitors and induced significantly different levels of consensus between groups. These results indicate that, while all groups value general management goals, groups that frequently travel the JIC place greater value on efficient and safe commutes.

Assessing the potential for conflict can help managers prioritize strategies that are most acceptable and least contentious, but this process can also help managers anticipate tradeoffs in cases where controversial strategies must be pursued in order to meet ecological goals (McShane et al., 2011; Vaske et al., 2010). For example, Crawford et al. (2014a) found that current rates of terrapin road mortality and raccoon predation of nests were sufficiently high that both must be addressed to stem the decline in the Jekyll Island terrapin population. In other words, reduction of raccoon predation on nests is needed to conserve terrapin populations; however, lethal raccoon removal was among the most contentious management options across patron groups. Therefore, managers might assume that general stakeholder support for terrapin management would translate to support for lethally removing raccoons if such action was biologically justified; however, our study shows that strong opposition to specific actions could compromise overall support for terrapin management. Measuring conflict and identifying unsupportive or contentious groups in these scenarios can identify needs for improved communication and education programs. In our study, conflicting attitudes may have arisen from patrons' lack of knowledge about the ecological impacts of problem species as well as management solutions, which was expressed by several patrons in written comments who requested more education. Many studies have found that knowledge of environmental threats or solutions is a significant antecedent of support for conservation strategies (Bremner & Park, 2007; Frick, Kaiser, & Wilson, 2004; Whittaker et al., 2001); hence, the development of outreach programs that emphasize raccoon threats to terrapins, especially targeted at residents and employees, may reduce conflict and increase support prior to management implementation.

JIA and GSTC researchers have begun testing at small scales on the causeway three strategies developed based on biological research (using hidden nest boxes, short fences, and flashing signage; Crawford et al., 2014a,b) that ranked highest in overall acceptability. Multiple strategies will likely be needed in order to have complementary impacts to road threats that stabilize at-risk terrapin populations (Crawford et al., 2014a; Maerz et al., *In review*). It is plausible that the combined impacts of these three strategies can sufficiently reduce road threats; nest boxes and fencing are designed to prevent terrapins from entering roads while also preventing raccoons from depredating nests laid inside boxes, and flashing signage can alert drivers across broader portions of the road when terrapin nesting activity is highest (Crawford et al., 2014b). These strategies have minimal expected impacts to driver safety, speed limits, or causeway aesthetics; thus, we anticipate minimal opposition to strategy implementation, especially if management is coupled with interpretive signage or local media releases that inform the public of the intended wildlife impacts. However, certain road management alternatives require driver behavior changes in order to be effective (e.g., speed reduction and flashing signage designed to increase awareness of animals on roads), so future research (following St

John et al., 2011) is still needed to evaluate the utility of road users' values, attitudes, and other psychosocial factors as predictors of behavior change that leads to threat reductions. Future research will monitor the effects of these strategies and identify the scales at which they should be applied to achieve population stability and growth. Given the widespread impacts of roads to wildlife and potential areas of stakeholder conflict, assessing the social dimensions surrounding road mortality is essential for developing biologically effective and socially acceptable road management and policy.

ACKNOWLEDGEMENTS

Funding to support this research was provided by the Daniel B. Warnell School of Forestry and Natural Resources, University of Georgia and Jekyll Island Foundation through an assistantship to B.A.C. We thank L. Larson, T. Norton, M. Kaylor, S. Mills, and D. Quinn for help in survey construction and project coordination. We thank associates of the Jekyll Island Authority and Jekyll Island Citizens Association for their continued collaborations. All methods were conducted with the approval of the University of Georgia Institutional Review Board (Project #: 2012-10815-1).

REFERENCES

Aipanjiguly, S., Jacobson, S. K., & Flamm, R. (2003). Conserving manatees: Knowledge, attitudes, and intentions of boaters in Tampa Bay, Florida. *Conservation Biology, 17,* 1098-1105.

Aresco, M. J. (2005). Mitigation measures to reduce highway mortality of turtles and other herpetofauna at a north Florida lake. *Journal of Wildlife Management, 69,* 549-560.

Baird, R. C. (2009). Coastal urbanization: The challenge of management lag. *Management of Environmental Quality: An International Journal, 20,* 371-382.

Bertwistle, J. (1999). *The effects of reduced speed zones on reducing bighorn sheep and elk collisions with vehicles on the Yellowhead Highway in Jasper National Park.* Paper presented at the the Proceedings of the International Conference on Wildlife Ecology and Transportation, Missoula, MT, September.

Boucquey, N., Campbell, L. M., Cumming, G., Meletis, Z. A., Norwood, C., & Stoll, J. (2012). Interpreting amenities, envisioning the future: Common ground and conflict in North Carolina’s rural coastal communities. *GeoJournal, 77,* 83-101.

Bremner, A., & Park, K. (2007). Public attitudes to the management of invasive non-native species in Scotland. *Biological Conservation, 139,* 306-314.

Buhlmann, K. A., & Osborn, C. P. (2011). Use of an artificial nesting mound by wood turtles (*Glyptemys insculpta*): A tool for turtle conservation. *Northeastern Naturalist, 18,* 315-334.

Bureau, U. S. C. (2010). *American Fact Finder*. Retrieved from http://factfinder2.census.gov/faces/nav/jsf/pages/community_facts.xhtml.

Clevenger, A. P., Chruszcz, B., & Gunson, K. E. (2001). Highway mitigation fencing reduces wildlife-vehicle collisions. *Wildlife Society Bulletin, 29*, 646-653.

Crawford, B. A., Maerz, J. C., Nibbelink, N. P., Buhlmann, K. A., & Norton, T. M. (2014a). Estimating the consequences of multiple threats and management strategies for semi-aquatic turtles. *Journal of Applied Ecology, 51*, 359-366.

Crawford, B. A., Maerz, J. C., Nibbelink, N. P., Buhlmann, K. A., Norton, T. M., & Albeke, S. E. (2014b). Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology, 51*, 367-375.

Dillman, D. A. (2007). *Mail and Internet surveys: The tailored design method* (2007 update, 2nd ed.). Hoboken, NJ: John Wiley & Sons.

Dodd, C. K., Barichivich, W. J., & Smith, L. L. (2004). Effectiveness of a barrier wall and culverts in reducing wildlife mortality on a heavily traveled highway in Florida. *Biological Conservation, 118*, 619-631.

Fahrig, L., & Rytwinski, T. (2009). Effects of roads on animal abundance: An empirical review and synthesis. *Ecology and Society, 14*(1), 21. http://www.ecologyandsociety.org/vol14/iss1/art21/

Feinberg, J. A., & Burke, R. L. (2003). Nesting ecology and predation of diamondback terrapins, *Malaclemys terrapin*, at Gateway National Recreation Area, New York. *Journal of Herpetology, 37*, 517-526.

Frick, J., Kaiser, F. G., & Wilson, M. (2004). Environmental knowledge and conservation behavior: Exploring prevalence and structure in a representative sample. *Personality and Individual Differences, 37*, 1597-1613.

Fulton, D. C., Manfredo, M. J., & Lipscomb, J. (1996). Wildlife value orientations: A conceptual and measurement approach. *Human Dimensions of Wildlife, 1*, 24-47.

Georgia Department of Transportation. (2011). *Georgia's State Traffic and Report Statistics (STARS)*. Retrieved from http://www.dot.ga.gov/informationcenter/statistics/stars/Pages/default.aspx.

Jacobson, S. K., & Marynowski, S. B. (1997). Public attitudes and knowledge about ecosystem management on Department of Defense land in Florida. *Conservation Biology, 11*, 770-781.

Karlsson, J., & Sjöström, M. (2007). Human attitudes towards wolves, a matter of distance. *Biological Conservation, 137*, 610-616.

Koval, M. H., & Mertig, A. G. (2004). Attitudes of the Michigan public and wildlife agency personnel toward lethal wildlife management. *Wildlife Society Bulletin, 32*, 232-243.

Maerz, J. C., Seigel, R. A., & Crawford, B. A. (*In review*). Terrapin conservation: Mitigating habitat loss, road mortality, and subsidized predators. In W. M. Roosenburg & V. S. Kennedy (Eds.), *The Ecology and Management of Diamondback Terrapins*. Baltimore, MD: Johns Hopkins Press, invited contribution: submitted.

Marcoux, A. (2005). *Deer-vehicle collisions: An understanding of accident characteristics and drivers' attitudes, awareness, and involvement.* (M.S. Thesis), Michigan State University, East Lansing, MI.

McKenzie-Mohr, D., Nemiroff, L. S., Beers, L., & Desmarais, S. (1995). Determinants of responsible environmental behavior. *Journal of Social Issues, 51*, 139-156.

McShane, T. O., Hirsch, P. D., Trung, T. C., Songorwa, A. N., Kinzig, A., Monteferri, B., . . . Pulgar-Vidal, M. (2011). Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation, 144,* 966-972.

Messmer, T. A., Brunson, M. W., Reiter, D., & Hewitt, D. G. (1999). United States public attitudes regarding predators and their management to enhance avian recruitment. *Wildlife Society Bulletin, 27*, 75-85.

Munscher, E. C., Kuhns, E. H., Cox, C. A., & Butler, J. A. (2012). Decreased nest mortality for the Carolina diamondback terrapin (*Malaclemys terrapin centrata*) following removal of raccoons (*Procyon lotor*) from a nesting beach in northeastern Florida. *Herpetological Conservation and Biology, 7*, 176-184.

Poudyal, N. C., Siry, J. P., & Bowker, J. (2010). Urban forests' potential to supply marketable carbon emission offsets: A survey of municipal governments in the United States. *Forest Policy and Economics, 12*, 432-438.

Riitters, K. H., & Wickham, J. D. (2003). How far to the nearest road? *Frontiers in Ecology and the Environment, 1*, 125-129.

Riley, S. J., Decker, D. J., Carpenter, L. H., Organ, J. F., Siemer, W. F., Mattfeld, G. F., & Parsons, G. (2002). The essence of wildlife management. *Wildlife Society Bulletin, 30,* 585-593.

Rosatte, R. (2000). Management of raccoons (*Procyon lotor*) in Ontario, Canada: Do human intervention and disease have significant impact on raccoon populations? *Mammalia, 64*, 369-390.

Sharp, R. L., Larson, L. R., & Green, G. T. (2011). Factors influencing public preferences for invasive alien species management. *Biological Conservation, 144*, 2097-2104.

Shindler, B., Brunson, M. W., & Cheek, K. A. (2004). Social acceptability in forest and range management. In M. J. Manfredo, J. J. Vaske, B. L. Bruyere, D. R. Field, & P. J. Brown (Eds.), *Society and Natural Resources: A Summary of Knowledge* (pp. 145-157). Jefferson, MO: Modern Litho Press.

Sokal, R., & Rohlf, F. (1995*). Biometry: The Principies and Practice of Statistics in Biological Research*. New York, NY: W. H. Freemann.

SPSS Inc. (2007). SPSS for Windows (Version 16.0). Chicago, IL: SPSS Inc.

St John, F. A., Edwards-Jones, G., & Jones, J. P. (2011). Conservation and human behaviour: Lessons from social psychology. *Wildlife Research, 37*, 658-667.

Stankey, G. H., & Shindler, B. (2006). Formation of social acceptability judgments and their implications for management of rare and little-known species. *Conservation Biology, 20*, 28-37.

Steel, B. S., List, P., & Shindler, B. (1994). Conflicting values about federal forests: A comparison of national and Oregon publics. *Society & Natural Resources, 7*, 137-153.

Stout, R. J., Knuth, B. A., & Curtis, P. D. (1997). Preferences of suburban landowners for deer management techniques: A step towards better communication. *Wildlife Society*, *25*, 348-359.

Sullivan, T. L., & Messmer, T. A. (2003). Perceptions of deer-vehicle collision management by state wildlife agency and department of transportation administrators. *Wildlife Society Bulletin*, *31*, 163-173.

Sullivan, T. L., Williams, A. F., Messmer, T. A., Hellinga, L. A., & Kyrychenko, S. Y. (2004). Effectiveness of temporary warning signs in reducing deer-vehicle collisions during mule deer migrations. *Wildlife Society Bulletin, 32*, 907-915.

Szerlag-Egger, S., & McRobert, S. P. (2007). Northern diamondback terrapin occurrence, movement, and nesting activity along a salt marsh access road. *Chelonian Conservation and Biology*, *6*, 295-301.

Thompson, S. C. G., & Barton, M. A. (1994). Ecocentric and anthropocentric attitudes toward the environment. *Journal of Environmental Psychology, 14*(2), 149-157.

Tavakol, M., & Dennick, R. (2011). Making sense of Cronbach's alpha. *International Journal of Medical Education, 2*, 53-55.

Trombulak, S. C., & Frissell, C. A. (2000). Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology, 14*, 18-30.

Vaske, J. J., Beaman, J., Barreto, H., & Shelby, L. B. (2010). An extension and further validation of the potential for conflict index. *Leisure Sciences, 32*, 240-254.

Vaske, J. J., & Donnelly, M. P. (1999). A value-attitude-behavior model predicting wildland preservation voting intentions. *Society & Natural Resources, 12*, 523-537.

Vaske, J. J., & Donnelly, M. P. (2007). Public knowledge and perceptions of the desert tortoise. *Report for the National Park Service* (HDNRU Report No. 81). Fort Collins, CO: Colorado State University, Human Dimensions in Natural Resources Unit.

Whittaker, D., Manfredo, M. J., Fix, P. J., Sinnott, R., Miller, S., & Vaske, J. J. (2001). Understanding beliefs and attitudes about an urban wildlife hunt near Anchorage, Alaska. *Wildlife Society Bulletin*, *29*, 1114-1124.

Zinn, H. C., Manfredo, M. J., Vaske, J. J., & Wittmann, K. (1998). Using normative beliefs to determine the acceptability of wildlife management actions. *Society & Natural Resources, 11*, 649-662.

Table 3.1. Descriptions and acceptability ratings among Jekyll Island patrons (*N* = 1,344) for potential management actions for reducing terrapin road mortality or nest predation on the Jekyll Island Causeway. Actions are ranked by mean ratings of the degree of acceptability measured on a 5-point response scale (-2 = Not at All Acceptable, 2 = Highly Acceptable).

Action	Intended effects (Source)	Mean (± SE) acceptability
General terrapin mgmt.	Maximize population persistence	1.36 (0.02)
Nest boxes hidden	Prevent terrapins from entering road, protect eggs from predation, lower impact to aesthetics (Buhlmann & Osborn, 2011)	1.14 (0.03)
Short fences	Prevent terrapins from entering road, lower impact to aesthetics (Aresco, 2005; Clevenger, Chruszcz, & Gunson, 2001)	1.05 (0.03)
Flashing signage	Increase driver awareness during peak terrapin activity (Sullivan et al., 2004)	0.99 (0.03)
Nest boxes in plain sight	Prevent terrapins from entering road, protect eggs from predation (Buhlmann & Osborn, 2011)	0.82 (0.03)
Speed reduction	Increase driver reaction time, increase travel time (Bertwistle, 1999)	0.43 (0.04)
Clear-cut hedges	Increase area of optimal nesting habitat, may reduce terrapins entering road, increase scenic views (Suggested in Crawford et al., 2014a; Szerlag-Egger & McRobert, 2007)	0.35 (0.04)
Relocate raccoons	Reduce nest predation rates, may be temporary due to raccoons returning (Rosatte, 2000)	0.14 (0.04)
Concrete barriers	Prevent terrapins from entering road (Dodd, Barichivich, & Smith, 2004)	-0.31 (0.04)
Speed bumps	Increase driver reaction time, increase travel time (Suggested in Clevenger et al., 2001)	-0.43 (0.04)
Lethally remove raccoons	Reduce nest predation rates (Munscher, Kuhns, Cox, & Butler, 2012)	-0.54 (0.04)
Do Nothing	-	-1.35 (0.03)

Table 3.2. Overall descriptive statistics and parameter estimates from the logistic regression model predicting terrapin management attitudes of Jekyll Island patrons (*N* = 1,344).

Variable	**Definition**	**Total N**	**Mean**	**SE**	**β**[c]	**Exp(β) odds ratio**
General mgmt attitude[a]	1 = attitude score of 1 or 2, 0 otherwise	1,332	.771	.012	Response variable	
Value orientation[b]	1 = value score of 1 or 2, 0 otherwise	1,301	.643	.013	1.863***	6.443
EXPNat	1 = visited Jekyll nature centers	1,342	.593	.013	0.375*	1.454
EXPTerps	1 = witnessed terrapin mortality	1,343	.218	.011	0.169	1.184
EXPVisits	1 = return visitor	1,311	.809	.011	-0.015	0.985
Age Dummy	1 = 40 or older	1,327	.596	.013	-0.525***	0.592
Gender Dummy	1 = female	1,330	.639	.013	0.534***	1.706
Ethnicity Dummy	1 = ethnic minority, 0 = white	1,320	.068	.007	0.068	1.070
HigherEd Dummy	1 = advanced degree	1,328	.580	.014	0.062*	1.332
Visitor		1,043			0.600*	1.822
Resident		214			0.807**	2.241
Employee		87			Comparison group	

[a] Composite score based on 3 items, Cronbach's alpha = .717

[b] Composite score based on 7 items (shared common factor from Principal Axis Factoring), Cronbach's alpha = .830

[c] Model Fit Statistics: Nagelkerke Pseudo R^2 = .259. Full model χ^2 (df = 10) = 230.3, $p < .001$. Classification accuracy (77.8%) exceeded proportional by chance accuracy rate (46.9%). Model constant = -1.630 (significant at $p < .01$)
* $p < .1$, ** $p < .05$, *** $p < .01$

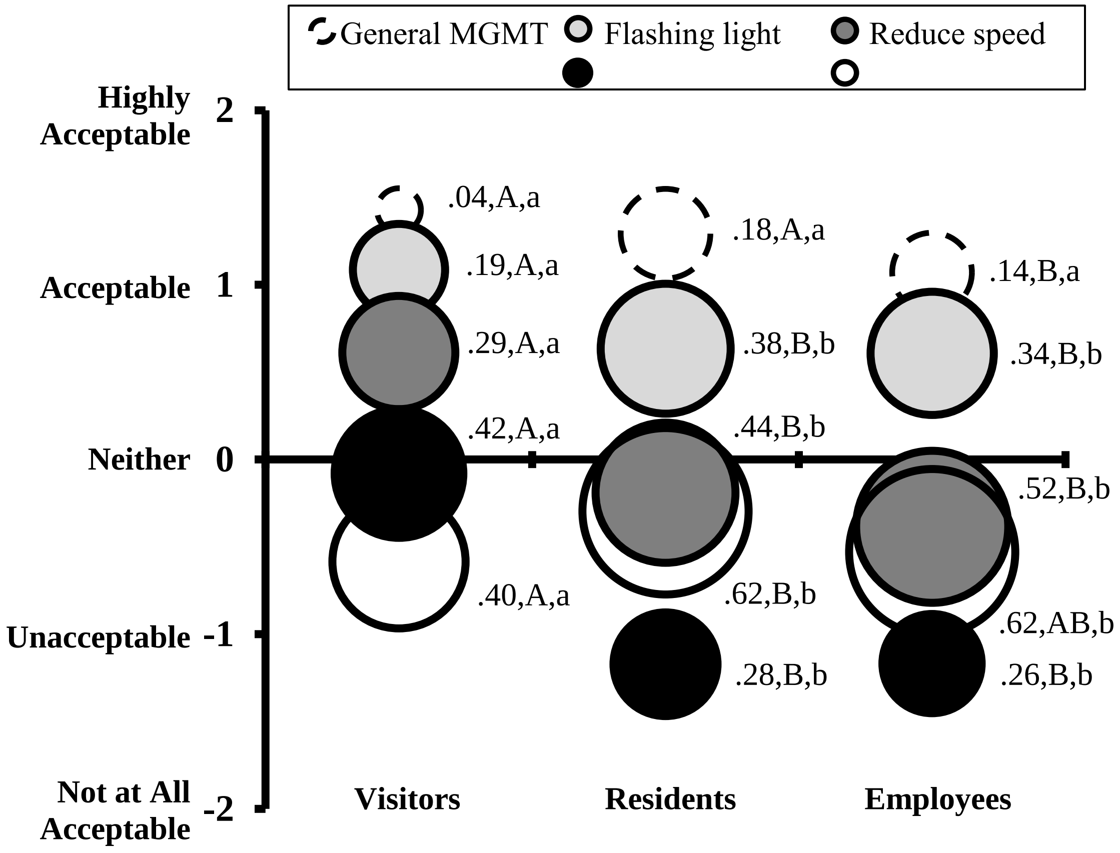


Figure 3.1. Mean response for the acceptability and the Potential for Conflict Index (PCI$_2$) for select terrapin-causeway management actions by patron groups of Jekyll Island, GA. Bubbles are centered on the mean acceptability response and the size of the bubble and the adjacent label is the PCI$_2$ value, which ranges from 0 (minimum conflict) to 1 (maximum conflict). Mean acceptability values (uppercase letters) and PCI$_2$ values (lowercase letters) differed by action ($F_{2,1302\text{-}1327}$ = 3.37-73.78, $p < .017$ Bonferroni-corrected), with different letters indicating significantly different groups for each management action.

CHAPTER 4

APPLYING AN INTEGRATED ANALYSIS TO EVALUATE MANAGEMENT DECISIONS FOR A DECLINING SPECIES[2]

---

[2] Crawford, B.A., C.T. Moore, T.M. Norton, and J.C. Maerz, To be submitted to *Ecological Applications*

ABSTRACT

A key challenge for making conservation decisions is predicting how wildlife populations respond to multiple, concurrent threats and potential management strategies, usually under substantial uncertainty. Obtaining accurate population predictions is contingent on robust estimation of demographic rates impacted by threats and management. Novel modeling approaches have been developed to produce more precise estimates even when datasets are limited or partial, which is typical for cryptic or rare species of conservation concern. The aim of this research was to apply an integrated modeling approach to inform management decisions within the context of multiple, road-associated threats impacting a declining species (the diamondback terrapin *Malaclemys terrapin*) on Jekyll Island, Georgia, USA. We implemented this approach in two phases: 1) we developed an integrated model to jointly estimate demographic rates from two mark-recapture datasets, while directly estimating the impact of management strategies (flashing warning signage) deployed during the study, and 2) we projected the population using population viability analysis under different scenarios representing no management, current actions, and potential strategies. Under no management, local population extirpation was nearly certain. We estimated significant increases in survival under current actions (i.e., when signage was implemented), but additional actions included in potential strategies were needed to further increase the probability of persistence and stabilize the terrapin population. Population persistence was most sensitive to the proportion of individuals crossing the road, and strategies that included extensive roadside barriers led to stable populations. However, best-performing strategies also included complementary actions (e.g., predator management to increase nest success), which supports previous conservation studies showing mitigation of multiple threats is required to ensure the viability of declining populations.

This work builds on previous research that used ecological patterns to identify management targets and precedes research that will incorporate population persistence estimates in the context of other socioeconomic objectives within a collaborative structured decision making process with local stakeholders.

INTRODUCTION

Conservation management is commissioned with addressing wicked problems, meaning those involving complex interactions between social and ecological systems, multiple, concurrent threats to natural resources, and nearly unlimited potential strategies whose outcomes are uncertain (Game *et al.* 2014). Concomitantly, ecological modelling is commissioned with inferring and predicting system dynamics, upon which management actions can be based. In order to form robust conservation decisions, modeling approaches should be developed that are realistic in the representation of context-specific processes and transparent in the treatment of key uncertainties (Schmolke *et al.* 2010).

Population viability analysis (PVA) is an effective tool for estimating outcomes of interest (e.g., population abundance, growth, persistence) for conservation decisions (Akçakaya & Sjögren-Gulve 2000; Morris & Doak 2002). PVAs iteratively project populations through time under various conditions, including current threats and potential management strategies, while accounting for the effects of environmental and demographic stochasticity (Akçakaya & Sjögren-Gulve 2000; McGowan, Runge & Larson 2011). PVA models use demographic rates to advance individuals within a population through ages, life stages (e.g., juvenile, adult), or states (e.g., breeder, non-breeder). Models then incorporate the effects of natural or anthropogenic stressors, as well as alteration of these stressors via management activities, on one or more subsets of the population (Morris & Doak 2002). Modeling multiple, concurrent threats within a single PVA is crucial for decision making, since unmitigated threats may render targeted management actions ineffective (Heppell, Crowder & Crouse 1996; Rhodes *et al.* 2011; Crawford *et al.* 2014a). PVAs are highly adaptable to a species' life history and socioecological contexts being modeled and can incorporate parameter estimates from different sources, such as empirical studies and expert opinion (Akçakaya & Sjögren-Gulve 2000).

Obtaining reliable predictions from PVAs remains challenging within many conservation contexts given multiple sources of uncertainty that impact the parameters used to project populations. Population parameters (e.g., initial abundance, survival, productivity) are estimated from observation data; thus, uncertainty around parameter estimates inherently includes variation of the demographic process as well as observation error that should be separated before making inferences (Clark & Bjørnstad 2004). Overestimation of demographic rates and increases in uncertainty can occur for species with limited available data or low detection (e.g., Zipkin *et al.* 2014). In some cases, there may be no current data on which to estimate parameters, such as for rare species or novel, untested management actions, and PVAs must rely on expert elicitation associated with higher degrees of uncertainty (Krueger *et al.* 2012). Furthermore, the impacts of management strategies are difficult to estimate for long-lived species with low productivity and delayed maturity since longer post-management periods are necessary to observe changes in population growth (Heppell, Crowder & Crouse 1996; Moore *et al.* 2012; Tempel, Peery & Gutiérrez 2014). Since PVAs are commonly applied to rare, declining, and cryptic species with sparse datasets, novel approaches have been developed to address these challenges for improved management decisions.

Joint modeling frameworks can leverage information contained in multiple, partial datasets to estimate shared demographic processes for a population, which increases precision, ensures consistency of estimates across datasets, and reduces effects of potential bias of individual datasets (Schaub & Abadi 2011). Examples of these frameworks include the joint live-dead encounter model for mark-recapture and dead-recovery data developed by Burnham (1993) and, more recently, integrated population models (IPMs) for the unified analysis of mark-recapture, population count, and other datasets (Schaub & Abadi 2011). Bayesian PVAs formally incorporate uncertainty around parameter estimates while separately modeling annual stochasticity in population simulations (Wade 2002; Moore *et al.* 2012), which reduces the risk of overestimating population persistence (McGowan, Runge & Larson 2011). Bayesian PVAs allow for efficient evaluation of the sensitivity of model outcomes to changes in parameter values, including those estimated by expert opinion (Wade 2002).

Diamondback terrapins (*Malaclemys terrapin*) inhabit salt marshes along the Eastern and Gulf Coasts of the United States – regions experiencing the fastest annual increases in developed area, road density, and traffic loads (Baird 2009). Multiple anthropogenic threats contribute to terrapin population declines (Roosenburg 1991; Gibbons *et al.* 2001; Grosse *et al.* 2011; Crawford *et al.* 2014a; Isdell *et al.* 2015; Chambers & Maerz *in press;* Maerz, Seigel & Crawford *in press*). Terrapins are frequent bycatch in commercial and recreational crab pot fisheries (Roosenburg *et al.* 1997; Grosse *et al.* 2011; Chambers & Maerz *in press*), and in areas where roads fragment salt marsh, adult females are struck by vehicles while searching for elevated nesting habitat each summer (Butler, Heinrich & Seigel 2006; Szerlag-Egger & McRobert 2007; Crawford *et al.* 2014b). Terrapins share characteristics with the majority of turtles (e.g., long-lived, delayed maturity, naturally high adult survival) that are likely to make populations susceptible to even low rates (3–10%) of additive mortality due to roads (Gibbs & Shriver 2002; Steen & Gibbs 2004; Butler, Heinrich & Seigel 2006; Maerz, Seigel & Crawford *in press*). Subsidized predators, such as raccoons *Procyon lotor*, contribute to high rates (50–90%) of nest mortality on roadsides and other developed areas (Crawford 2015; Maerz, Seigel & Crawford *in press*). The density of roadside vegetation can also influence terrapin demographic rates. Grosse *et al.* (2015) observed higher predation rates and higher proportions of male hatchlings for nests laid in man-made hedgerows (commonly cedar and wax myrtle *Myrica cerifera*), relative to cleared, open areas along roadsides. Like many reptiles, terrapins exhibit environmental sex determination (ESD) where warmer incubation temperatures produce greater proportions of female offspring (Ewert, Jackson & Nelson 1994). While existing management practices have targeted road mortality (Aresco 2005b) and predation (Munscher *et al.* 2012), vegetation management practices also have the potential to increase population growth (Maerz, Seigel & Crawford *in press*).

OBJECTIVES

The aim of this research was to apply an integrated modeling approach to evaluate the consequences of management strategies to inform decision making within the context of road impacts on wildlife. We used a population of terrapins that nest on the causeway to Jekyll Island, Georgia, USA as a model system. Our specific objectives were (i) to develop an integrated model to jointly estimate demographic rates from two mark-recapture datasets, (ii) to construct a stage-based population model incorporating demographic rates estimated in this and other studies and from expert opinion, and (iii) to project population growth and persistence under different scenarios representing no management, current actions, and potential strategies to reduce road-associated threats. This work builds on previous research that estimated the effects of road-associated threats, predicted local population declines in the absence of management, and identified management targets (Crawford *et al.* 2014a; Crawford *et al.* 2014b; Grosse *et al.* 2015). In response to this information, terrapin experts, local stakeholders, and the Jekyll Island State Park Authority, the agency commissioned with managing the area, have specified potential management strategies as part of a collaborative structured decision making process. This study precedes research that will incorporate population persistence outcomes for each strategy in the context of other socioeconomic objectives for management of the JIC.

METHODS

Study area and population

We conducted research in conjunction with long-term monitoring efforts of the Georgia Sea Turtle Center (GSTC) on the 8.7-km Downing-Musgrove Causeway to Jekyll Island, GA, USA (31.08°N, 81.47°W; Figure 4.1). The Downing-Musgrove Causeway (aka Jekyll Island Causeway: JIC) is a state highway with average annual daily traffic (AADT) of 4,020 vehicles day^{-1} (Georgia Department of Transportation 2014). The JIC bisects a salt marsh peninsula consisting of a network of intertidal creeks and high marsh dominated by *Spartina spp*. We define the population of interest for this study as terrapins inhabiting this peninsula and using JIC roadsides for nesting, but we allow for permanent and temporary emigration out of the study site in the integrated model. The JIC represents a regional road mortality hot spot where 100–400 adult female terrapins are killed each summer while in search of roadside nesting habitat (Crawford *et al.* 2014b; GSTC, unpubl. data). Early monitoring of terrapin nesting seasons on the JIC revealed that crossing activity was concentrated spatially on road sections (hot spots) and temporally within a daily 3-hr period around the scheduled diurnal high tide (Crawford *et al.* 2014b), and we used these peaks to specify targets for management during this study. While the majority of management actions evaluated in this study were proposed by stakeholders and have yet to be tested on the JIC, we have deployed two actions (Figure 4.2). In 2011, we constructed a 22-m hybrid barrier composed of nest boxes at one road crossing hot spot and measured the reduction in terrapins on the road at this site (Quinn *et al.* 2015; Maerz, Seigel & Crawford *in press*). Nest boxes consisted of elevated man-made mounds of sand with electrified cages on top, which were designed to allow terrapins access to the box while excluding mammalian and avian predators (see Buhlmann & Osborn 2011). In 2013, we collaborated with the Georgia Department of Transportation to install two terrapin crossing signs with flashing warning beacons (R829 Series Solar School Zone Flashing Beacons, Carmanah Technologies®, Victoria, British Columbia, Canada) to alert drivers entering a 6-km section of the JIC from either direction (Maerz, Seigel & Crawford *in press*). We activated signs for 2 hrs per day during the temporal peak of terrapin crossing around the scheduled diurnal high tide (Crawford *et al.* 2014b).

Data collection

We collected two datasets from the terrapin population (Figure 4.3) to develop the integrated model: 1) a mark-recapture dataset of adult male and female terrapins inhabiting two creeks adjacent to the JIC from 2010 to 2015, following Pollock's Robust Design (Pollock 1982), and 2) a multistate mark-recapture-recover dataset of adult nesting females on the JIC from 2009 and 2015, following a joint live-dead encounter design (Burnham 1993).

At a local scale of two creeks adjacent to the JIC, we sampled for three consecutive days (secondary periods) each April (primary periods) from 2010 to 2015 (18 total capture occasions) under a Robust Design. Separate teams seined an 800-m section of each creek concurrently, using mesh seines following the methods of Dorcas, Willson and Gibbons (2007). Captured animals were sexed based on body size, the position of the cloaca on the tail, and head allometry. All were measured for shell length, and unmarked animals were given a unique code by drilling or notching marginal scutes. We released all animals prior to the next sampling occasion. Seining equipment allowed for the capture of adult and juvenile stages of both sexes known to use creeks (Dorcas, Willson & Gibbons 2007; Grosse *et al.* 2011), but we only captured 18 individuals

(6.7% of all captures) identified as juveniles based on shell lengths smaller than mature terrapins (Gibbons *et al.* 2001). Therefore, we censored our dataset to include only adult males (294 encounters of 194 individuals) and females (68 encounters of 56 individuals). Since we only captured 33 terrapins (with 7 total recaptures) from one of the creeks during the study period, we combined data from both creeks for analysis.

At the larger population scale for our study, we performed capture-mark-recovery of live and dead adult females attempting to nest on the JIC. We conducted intensive road surveys of the JIC during seven consecutive nesting seasons (1 May – 15 July) from 2009 to 2015. We drove along the JIC every 20–90 mins between 0800 to 2000 each day during the study period. We observed and recorded terrapins in three states: alive and crossing (AC), alive and not crossing (ANC), and dead on the road (DC). Since both roadsides are suitable nesting habitat, some terrapins will emerge from the marsh and nest on the proximate roadside without attempting to cross. Alternatively, terrapins may attempt to cross, nest on the opposite roadside, and then attempt a return trip across the road. We recorded any terrapin observed on the road surface or within 1 m of the road and walking toward it as AC, and we intervened by capturing these individuals before they could be struck by vehicles. If we observed a terrapin that nested and attempted to return to the adjacent marsh without crossing the road, we recorded it as ANC. These behaviors were consistently observed in the field, so we were confident in our ability to assign a terrapin to each state. We recorded any terrapin, dead or still alive, found struck by a vehicle on the road as DC. We processed and marked all uninjured terrapins using the same methods described above, and we placed animals in nest boxes located roughly every mile on both sides of the road within 1 hr of capture. Injured or dead terrapins were taken to the GSTC to confirm identity if marked and to receive veterinary care. We recognize that our capturing of AC individuals likely affected their fate when vehicles were nearby. Therefore, the estimated effect of road mortality on survival rates of AC females is likely conservative (see Results). The final dataset contained 2307 encounters (AC = 1065, ANC = 227, DC = 1015) of 1984 individuals.

Parameter estimation

Although our datasets differed in individuals targeted within the population and scale of inference, they were assumed to share underlying demographic processes for females. Thus, we developed a state-space integrated model to jointly estimate population parameters of interest under a single model fitted in a Bayesian framework. This approach parallels that of IPMs (Schaub & Abadi 2011); however, IPMs typically include population count data and project changes in population size that result from estimates of demographic parameters. Alternatively, we used intensive mark-recapture datasets but not population count data, and we performed estimation and projection of the terrapin population in two separate steps instead of under a unified model. Our motivation for this approach was to allow for the inclusion of greater complexity in the projection model, where we incorporated estimates from the integrated model, previous studies, and expert opinion, while avoiding lengthy computer processing time expected if we merged the estimation and projection models.

The integrated model consisted of two submodels with independent and shared parameters: 1) a Robust Design model was fit to the creek mark-recapture data (m_{RD}) and 2) a multistate parameterization of a Jolly-Seber model (Kéry & Schaub 2012) was fit to road live-recapture-dead-recovery data (m_{MS}) (Figure 4.3). Notably, the parameters shared between submodels and informed by both datasets were female survival (φ), entry (b), and site fidelity (γ). Following Kéry and Schaub (2012), the entry probability in our model represents the process of entering the population either via recruitment from surviving and graduating from the juvenile stage within the population or immigration from another population. In the absence of other information, we assumed the rate that terrapins temporarily move in and out of the sampling area to be equal, irrespective of an animal's previous availability state; therefore, we used the single parameter γ to represent site fidelity (remaining in the study area). We present the comprehensive development of each submodel in Appendix B but illustrate core components below. Using Deviance Information Criterion (DIC: Spiegelhalter *et al.* 2002), we compared *a priori* models within a candidate set for the Robust Design submodel where parameters differed by sex, time, and covariate effects (see Appendix B). We then selected the parameterizations most supported by the data to use in the final integrated model. We did not perform model selection for the multistate submodel since it included a larger set of parameters, and preliminary model iterations showed lack of convergence when additional temporal effects were included for nuisance parameters. In the absence of other information, we used the simplest model, setting these parameters constant across years, to facilitate model convergence and estimation of target parameters.

Robust Design model

We developed a Robust Design model to estimate demographic rates and derive local abundances for male and female terrapins. The model was fit to individual encounter histories, where 1 denoted the individual was captured, 0 otherwise, for the 18 sampling events. Typical of Robust Design models (Pollock 1982), we assumed the population was closed to survival, entry, and site fidelity during the three-day secondary periods (t: days) but open between primary periods years (y: years).

Given low recapture rates in our study and the potential temporary movement of terrapins in and out of the sampling area, we constructed the submodel using additional parameters to formally account for zero-inflation in our dataset. We fit the submodel for Robust Design data using parameter-expanded data augmentation (Royle & Dorazio 2012), where a large number of all-zero capture histories (representing pseudo-individuals) were added to the dataset. Following Kéry and Schaub (2012), our "superpopulation" model first assigned an inclusion state (w_i) for each individual *i* in the augmented dataset that is drawn from a Bernoulli trial with probability ψ_{RD}, where $w_i = 1$ if the individual is a member of the superpopulation and 0 otherwise. Thus, the superpopulation size represents the total number of individuals that were alive and in the population at any time during the study. Next, we modeled the state history ($z_{i,y}$) for individuals describing whether the individual was alive and in the population ($z_{i,y} = 1$, 0 otherwise) each year as a Bernoulli trial with probability $b_{g,y}$, given that the individual of sex *g* had not yet entered the population, or $\varphi_{g,y}$, given that the individual had previously entered the population. We derived estimates for local abundance ($N.loc_{g,y}$) of males and females by summing the z states for all individuals in the superpopulation for each sex and each year. We modeled site fidelity, the process of an individual being available for sampling (i.e., inside the sampling area), each year as a Bernoulli trial with probability $\gamma_{g,y}$. We modeled the observation process of capturing a terrapin each sampling occasion, given that it is a real member of the superpopulation, alive in the population, and available for capture that year, as a Bernoulli trial with probability $p_{g,t}$. We found no evidence of behavioral effects from capture (see Appendix B) and set capture and recapture parameters as equal.

We modeled survival as a fixed effect for sex, a random year effect for each sex drawn from a zero-centered normal distribution with variance parameter σ_g^2, and an additional fixed effect (β_{per}) for the mean effect of period (before-after) relative to road management (flashing sign installation) on female survival. β_{per} was governed by an indicator variable (0 or 1) that allowed it to be included in the estimation of survival only in years after management was initiated. We modeled the process of annual entry into the population as a vector of probabilities that sums to 1 using a Dirichlet distribution constructed from prior random variables drawn from an uninformative gamma distribution. We then re-expressed these as conditional entry probabilities, the probability of entry at year *y* given that the individual had not yet entered the population (see Kéry & Schaub 2012), for each year and sex. We modeled site fidelity with a fixed sex and random year effect drawn from a zero-centered normal distribution with variance parameter σ_{F}^2. Lastly, we modeled capture as a fixed effect for sex, a random day effect drawn from a zero-centered normal distribution with variance parameter σ_{P}^2, and fixed effects for tide amplitude β_{tide} (obtained from http://tidesandcurrents.noaa.gov). Tide amplitude may negatively affect detection since higher tides create wider, deeper creeks that increases the chance a terrapin can evade the fixed-length seines we used.

Multistate mark-recapture model

We developed a multistate mark-recapture submodel that combined data on recaptures of live individuals and recoveries of dead individuals on the road to estimate survival and abundance of adult females in the JIC population. We considered individuals to be in one of six true states each year in the model: not yet entered the population (NYE), alive and crossing the JIC (AC), alive and not crossing the JIC (ANC), alive and outside (i.e., not nesting on) the JIC (AO), dead on the road (DC), and dead elsewhere (DO). We recorded individuals in the dataset as being in one of four observation states each year that were the outcome of underlying states: alive and crossing (AC), alive and not crossing (ANC), dead on the road (DC), and not seen (NS).

Using the same approach as before, we fit the multistate submodel to the dataset augmented with capture histories of all NSs, and we assigned an inclusion state (w_i) for each individual in the augmented dataset that is drawn from a Bernoulli trial with probability ψ_{MS}. Next, we modeled the individual's state histories ($z_{i,y}$) each year using a matrix of conditional probabilities of being in a particular state given the state in the previous year. The full matrix of conditional probabilities is presented in Appendix B. Briefly, the parameters governing these state conditional probabilities were as follows: given that an individual has not entered the population before year y, it may enter with probability b_y; it may remain in the study area and use the road for nesting (γ_y); and given that an individual is nesting in the study area, it may cross the road with probability c and survive either having crossed the road ($\varphi.c_y$) or not crossed the road ($\varphi.nc_y$). We modeled the process of observing a terrapin in each sampling year, given that it is a real member of the superpopulation, using a matrix of conditional probabilities (Appendix B), which included the probability of capturing a live terrapin, given that it was crossing ($p.c$) or not crossing ($p.nc$), and the probability of recovering an individual given that it died on the road (r). We considered any individual that had not yet entered the population, was alive but outside the study area, or died anywhere off the road (from natural or human causes) as unobservable. We derived estimates for population abundance ($N.JIC_y$) by summing the z states for all individuals in the superpopulation that were crossing, not crossing, or temporarily outside the study area each year.

In addition to parameters governing survival in the Robust Design submodel, we estimated specific survival of crossing and non-crossing females using a zero-sum fixed effect (β_{cross}) for crossing where the same value was added to the survival of non-crossers and subtracted from survival of crossers. We specified an independent fixed effect (β_{int}) for crossing and non-crossing females to represent the interaction between crossing status and management period. This effect was governed by the same indicator variable as β_{per} that allowed it to be included in the estimation of survival only in years after management was initiated, and thus, allowed the estimation of the impact of flashing signs on survival of crossing females relative to non-crossing females and years before installation. Entry probability and site fidelity shared effects for females between submodels. We set observation parameters (*p.c*, *p.nc*, and *r*) as constant across years.

We performed Bayesian analyses of both submodels and the integrated model using Markov chain Monte Carlo (MCMC) methods in Jags called from R (R Core Team 2013) via the R2jags package (Su & Yajima 2012). We assigned uninformative prior distributions for all fixed effect parameters and hyperparameters governing random effects in all models. For the integrated model, we estimated posterior distributions using 60000 iterations of three chains after discarding the first 50000. We retained all iterations (i.e., did not “thin” chains), which is recommended to improve computational efficiency without loss of estimation precision (Link & Eaton 2012), yielding a final set of 30000 samples from posterior distributions of the parameters. We assessed convergence for all models by visually inspecting chain mixing in MCMC trace plots and posterior distribution plots for evidence of unimodality and by calculating the Brooks-Gelman-Rubin diagnostic (Brooks & Gelman 1998), which compares within- and between-chain variance. We based parameter inferences on posterior means and 95% Bayesian credible intervals (BCIs; 2.5th – 97.5th percentile of the distribution). Our R and JAGS code is in Appendix C.

PVA model

We constructed a stage-based PVA model for the JIC terrapin population (Figure 4.4) and predicted population growth and persistence probability under baseline conditions, current management, and 19 potential management scenarios. We ran 1000 iterations of each scenario and projected the population for 50 years. To compare the consequences of management strategies, we estimated persistence probability as the proportion of iterations (out of 1000) that did not reach a quasi-extinction threshold of 50 individuals during the 50-year time horizon. We also estimated the mean population growth rate (λ) for each scenario by averaging the annual growth rates across the final 30 years of all iterations, where the annual growth rate was calculated as N_{y+1}/N_y. We discuss the construction of the baseline PVA model first, followed by alterations made to reflect management scenarios.

Because we know a great deal about terrapin life history, behavior, and impacts of anthropogenic threats related to roads from this and other studies, we expanded simple stage-based models (Lefkovitch 1965; Morris & Doak 2002) to represent dynamics realistic for the JIC population. We included males and females in our model given differing sex-specific rates of maturity and anthropogenic threats (i.e., road mortality). We grouped individuals into six stage classes based on previous life history and population modeling studies (Lovich & Gibbons 1990; Gilliand, Chambers & LaMar 2014): female hatchlings (X_h; age 1), female juveniles (X_j; ages 2-5), female adults (X_a; ages 6+), male hatchlings (Y_h; age 1), male juveniles (Y_j; ages 2-4), and male adults (Y_a; ages 5+). The model used a one-year time step and advanced individuals in each life stage through stochastic processes of survival, reproduction, and transition among stage classes using appropriate distributions (i.e., Binomial, Multinomial, Poisson) and demographic rates estimated from the integrated model or obtained from the literature (Table 4.1). Since the juvenile stage lasted multiple years, we separated annual juvenile survival into the probability of surviving and remaining a juvenile (P_j) and the probability of surviving and graduating to the adult stage (G_j) following methods of Caswell (1989). We did not include a maximum age or density-dependent effects in this model.

Our model recognized additional steps related to survival of different subclasses of adult females and recruitment of hatchlings from different habitats on the JIC. We divided the adult female stage class into three subclasses: individuals nesting on the JIC and crossing the road ($X_{a.c}$), individuals nesting on the JIC but not crossing the road ($X_{a.nc}$), and individuals not nesting on the JIC ($X_{a.o}$). Individuals entering or remaining in the adult female stage class were assigned to each subclass with probabilities of $\gamma * c$, $\gamma(1-c)$, and $1-\gamma$, respectively (parameters described above). Crossing females survived with probability $\varphi.c$ while the other two subclasses of adult females survived with probability $\varphi.nc$. To model the processes of annual fertility (F) and hatchling sex-determination (h) that yield the number of individuals that are born and survive to age 1, we specified the following sequential steps: adult females that survived and nested on the JIC were assigned to one of three habitats (hedge, open, or nest boxes) in proportion to the availability of each type (i.e., we did not assume any bias toward selecting a certain habitat type); nests survived (i.e., were not depredated) based on habitat-specific rates of nest success (ns_{hab}); eggs survived to produce new age 1 individuals with probability $\varphi_e * \varphi_h^{(3/4)}$, which represents the probability of surviving the 3-month period during the egg stage and the remaining ¾ of the year as a hatchling; and hatchlings were assigned a sex using habitat-specific hatchling sex ratios (h_{hab}: the probability of a hatchling being female). We note that only adult females using the JIC contributed to recruitment of hatchlings each year; thus, we assumed individuals not using the JIC were either not breeding or nesting in an area outside the extent of our population of interest.

We parameterized the baseline PVA model to allow certain demographic rates to vary between iterations and/or years while other rates were kept constant. We used the same initial stage abundances for all iterations. We set the initial abundance of adult female subclasses as the mean annual abundances estimated from the integrated model. To calculate the initial abundance of adult males at the population scale, we 1) assumed the ratio of males to females estimated at the local (creek) scale was no different than the sex ratio at the population (JIC) scale, and 2) multiplied the local ratio by the total female population abundance. We calculated initial abundances for all other stage classes using the adult male and female abundances and the stable stage distribution found in another terrapin population (Gilliand, Chambers & LaMar 2014). We accounted for uncertainty due to parameter estimation and stochastic, year-to-year variation in the PVA model (Wade 2002; Moore *et al.* 2012). For each model iteration, we sampled, with replacement, mean adult survival rates representing baseline conditions (i.e., survival in years before JIC management was initiated) and a standard deviation of temporal variability for males, crossing females, and non-crossing females from their posterior distributions in the integrated model. The parameter values for each iteration defined probability distributions from which stochastic survival rates were drawn each year in the simulation. We sampled a mean probability of crossing and a mean probability of site fidelity from their posterior distributions for each year of each iteration of the baseline model. We kept all other demographic rates constant across iterations and years.

We altered parameters in the baseline PVA model to simulate current management actions and 19 alternative strategies created by stakeholders in a structured decision making workshop (see Chapter 5). Given the multiple road-associated threats impacting this and other terrapin populations, management strategies were composed of different combinations of actions representing four types of management expected to affect model parameters (Figure 4.4): 1) on-road signage, speed reductions, and awareness campaigns designed to improve survival of crossing females, 2) roadside barriers designed to reduce the proportion of nesting females crossing the road, 3) predator removal to improve nest success, and 4) roadside vegetation management to improve nest success and proportion of hatchlings born female. For example, one strategy proposed the installation of short fences along the entire JIC, no change in speed limit or road signs, and no predator removal; another strategy proposed installation of short fences only at hot spots of terrapin activity, reduction of speed limit, and culling of raccoons at hot spots.

We obtained estimates for management effects from this and other empirical studies or expert elicitation of 12 terrapin research professionals (Appendix B). For current and potential strategies that included the installation of flashing signage, we used the mean $\varphi.c$ estimated in years post-management from the integrated model, and we employed the same procedures of sampling from posterior distributions to account for parameter uncertainty and stochasticity. We altered the mean survival rate for strategies that included additional signage, speed reductions, or education campaigns using estimated effects from expert opinion. We estimated habitat-specific rates of nest success and hatchling sex ratios under the current management strategy through empirical research and monitoring (Grosse *et al.* 2015), and we altered these rates for potential strategies using estimates from expert elicitation. Finally, we estimated the effects of predator management on nest success via expert opinion. See Appendix B for further details about incorporating management effects into PVA model scenarios and Appendix C for our R code for the PVA model.

Sensitivity analysis

We explored the sensitivity of mean population growth rate and persistence to variation of model parameters in two stages: 1) we altered demographic rates within the baseline PVA model, and 2) we altered expert estimates of management effects across the 19 potential management scenarios. First, we increased and decreased baseline means of the following demographic parameters one at a time by 5% with all other parameters unaltered: annual survival of each life stage and adult subclass (crossing and non-crossing females), site fidelity, and the probability that a nesting female will cross the JIC. Varying demographic rates by 5% represented small, realistic changes to parameters (less than the standard deviation for most estimates) and allowed for direct comparison of model outcomes. We explored baseline model sensitivity further by replacing the juvenile graduation probability with the mean entry probability for adult males and females, in turn, that were estimated from the integrated model. Entry probability represents a combination of processes of immigration from other sites and recruitment of adults from the juvenile class. Given that the JIC represents the majority of local nesting habitat (pers. obs.), movements over long distances (> 1km) are infrequently observed (Gibbons *et al.* 2001; Sheridan *et al.* 2010), and terrapins exhibit a high degree of creek and nest site fidelity (Tucker, Gibbons & Greene 2001; Sheridan *et al.* 2010; Crawford *et al.* 2014b), the rate of immigration relative to internal recruitment is likely small. We altered habitat-specific hatchling sex ratios in the baseline model. The initial model used relatively high habitat-specific hatchling sex ratios (proportion of females from nests in hedges = 15%, open and nest boxes = 100%) that were estimated by a single study, but the proportion of hatchling females from a nest may vary greatly with local climate and soil characteristics that influence development (Weisrock & Janzen 1999). Therefore, we replaced the original estimates with lower proportions of females (hedges = 0%, open and nest boxes = 51%) estimated by Weisrock and Janzen (1999) in an empirical study of another Emydid turtle species (C*hrysemys picta*). We estimated the change in mean λ resulting from 1000 runs of each model perturbation, relative to the baseline model. Second, we estimated mean persistence across management strategies by replacing mean estimates of management effects with the lower and upper 90% confidence intervals around estimates obtained from expert opinion, in turn, while keeping all other rates unaltered. We evaluated model sensitivity to the following management effects: survival of crossing females, barrier efficacy (proportion of females prevented from entering road), and nest success.

RESULTS

Annual captures varied considerably between years for both datasets. From the Robust Design dataset, individuals captured annually ranged from 4 to 100 males and 2 to 21 females. The annual number of terrapins observed in any state on the JIC ranged from 171 to 448. Of 1076 individual females marked during road surveying (crossing or not crossing), we recovered 107 struck and killed on the JIC. We recovered an additional 908 unmarked individuals dead on the road during the study period. The annual number of dead individuals recovered on the road ranged from 81 to 196.

At the local scale, we estimated a mean (95% BCI) annual abundance of 207 (125–303) males and 53 (33–80) females, yielding an estimated mean sex ratio of 80% males to 20% females. At the population scale, mean female annual abundance was 1684 (881–2760), which included individuals outside the study area. An estimated 1265 (788–1918) females used the JIC for nesting (individuals in AC, ANC, DC states) each year, on average. We present posterior mean estimates and 95% BCIs for target parameters used in PVA models in Table 4.1. The estimated mean survival rate was similar for males and females with overlapping BCIs. Female survival showed an interaction effect between crossing status and management implementation (Figure 4.5). Survival of crossing females was lower than non-crossing females in years pre- and post-management, but crossing female survival increased following management implementation while survival of non-crossing females remained similar. See Appendix B for posterior estimates of additional parameters.

PVA simulations under baseline (no management) conditions predicted a declining population (Figure 4.6a) with the lowest persistence probability (Figure 4.6b) among all scenarios. Current management conditions increased λ and persistence probability, but the population was still predicted to decline. Mean estimates of λ from the remaining 19 potential management scenarios ranged from 0.88 to 1.04 and persistence that ranged from 0.37 to 0.96. The persistence probability was high (>80%) for 8 strategies, each of which included extensive roadside barriers and predator removal as management actions. Four potential management strategies resulted in persistence probabilities <50%, and none included the use of extensive barriers or predator removal at the largest scale.

Sensitivity analysis revealed that λ was influenced most by proportional changes to adult female survival, followed by juvenile female survival and probability of crossing the road, and was influenced least by changes to site fidelity and juvenile and adult male survival (Table 4.2). Persistence probability changed marginally (< 10%) when we replaced mean estimates obtained from expert opinion with values representing lower and upper 90% confidence intervals for the effect of management on demographic rates, and the relative rank of management strategies to one another, measured by persistence probabilities, was insensitive to parameter uncertainty (Table 4.3).

DISCUSSION

Conducting an integrated analysis to estimate demographic rates and predict population trends for diamondback terrapins around Jekyll Island revealed findings essential for conservation decision making: (i) the population is predicted to decline with a low probability of persistence in the absence of additional mitigation of road-associated threats, and (ii) potential management strategies varied in their expected outcomes, which allows for the prioritization of strategies when considering the goal to maximize population growth and persistence. There are few studies to date that used joint analyses, such as integrated population models, of multiple data sources to explore the conservation status and management targets for a population (but see Rhodes *et al.* 2011; Tempel, Peery & Gutiérrez 2014). Our work contributes another example of applying integrated modeling approaches to robustly estimate population parameters within conservation contexts, and it is the first to evaluate impacts of management strategies in a decision making context. Population outcomes (e.g., persistence probability), often used to guide management decisions, hinge on the accuracy and precision of parameters used in predictive models. Thus, integrated models represent a valuable tool for future conservation efforts that can reduce uncertainty and bias around parameter estimates and resulting population predictions. Efficiently using all available data in a joint modeling framework is especially appropriate for data-limited species (e.g. rare or cryptic species), which are often the focus of conservation efforts. Numerous integrated models have been developed recently that can be adapted to various types of data, limitations of sampling designs, and context-specific demographic processes (reviewed in Schaub & Abadi 2011).

Although integrated analyses have been shown to increase the precision around parameter estimates compared to what is achievable by analyses of single datasets, a high degree of uncertainty remained for many estimates in our study. Low rates of detection across years in both mark-recapture datasets likely contributed to this low precision. Including parametric uncertainty in PVA is important for producing unbiased predictions (Wade 2002; Moore *et al.* 2012); however, this practice decreased the precision of population growth and persistence estimates from PVA simulations, especially attributable to uncertainty around parameters to which model outcomes are more sensitive. For example, previous studies of turtle population viability have shown that decreasing adult survival by as little as 3% can lead to declines (Congdon, Dunham & van Loben Sels 1993; Crawford *et al.* 2014a). We observed decreases in λ given a 5% reduction in adult female survival. . The 95% BCI for adult female survival included a range of 30% (range = 0.55 to 0.85) in our study, far exceeding levels that would impact population growth and persistence. While these results emphasize the need for further data collection to inform the estimation of sensitive parameters, our approach ensured that current uncertainties of terrapin population dynamics have been incorporated fully into final predictions. This transparency is paramount for evaluating the impacts of management strategies in a broader decision making framework.

As a species of conservation concern, diamondback terrapins are well-studied in human-impacted landscapes. Still, estimating certain demographic rates remains challenging given cryptic life stages (e.g., hatchlings) and habitats (e.g., use of extensive salt marsh) that limit rates of detection, long generation times that require longer study durations to measure population trends, and dynamic movement through complex habitats that impacts assumptions of population closure and subsequent inferences of survival and emigration. Our use of an integrated analysis advances previous modeling efforts for terrapins and provides novel insights into their population dynamics while also corroborating previous demographic estimates. Our estimates of survival for non-crossing adult females likely represent natural survival rates, consistent with previous estimates (Mitro 2003; Hart 2005), since no other direct female-specific threat exists in our study site. Mean male survival was slightly lower and more imprecise than previous estimates (Tucker, Gibbons & Greene 2001; Hart 2005). Our models accounted for demographic and observation processes related to low detection, but sparse capture and recapture data may have allowed some degree of bias and uncertainty to remain for estimates of male demographic rates. Conversely, estimates of demographic rates for females, for which we also had low detection in the Robust Design dataset, were informed by additional data from road surveying. The increased precision around female, relative to male, demographic rates illustrates the advantage of leveraging information in an integrated analysis when multiple, overlapping datasets are available. We must use caution in interpreting our estimates of site fidelity, since this parameter in our joint model represented fidelity to tidal creeks and, for females, fidelity to creeks and upland nesting sites (i.e., the JIC). We estimated higher site fidelity for males than females, which is consistent with findings that females were more likely to move between tidal creeks, presumably related to upland nesting movements (Gibbons *et al.* 2001; Tucker, Gibbons & Greene 2001; Sheridan *et al.* 2010; Maerz, Seigel & Crawford *in press*). For females, site fidelity was still high (0.683), which is consistent with previous findings that female terrapins exhibit high fidelity to nesting sites between years with few individuals dispersing to additional nesting sites (Szerlag-Egger & McRobert 2007; Sheridan *et al.* 2010; Crawford *et al.* 2014b). No study has estimated recruitment rates for terrapins, and we were only able to estimate entry probabilities, which combined processes of immigration and within-population recruitment. Since estimated λ was sensitive to juvenile survival and recruitment into the adult stage class, developing sampling and modeling techniques for estimating recruitment and immigration will inform future models of population dynamics.

Despite extensive work in the field of road ecology, evaluating population-level impacts of roads on wildlife remains challenging. Previous road mortality studies have assessed impacts using naïve counts of dead individuals (e.g., Aresco 2005a; Langen *et al.* 2007), predictive spatial models for wildlife-vehicle collisions based on species and road characteristics (e.g., Hels & Buchwald 2001), survival estimates from limited mark-recapture or radio-telemetry (e.g., Mumme *et al.* 2000; Row, Blouin-Demers & Weatherhead 2007), or indirect indicators (e.g., population density, sex ratio) of road mortality in sites of varying road densities (e.g., Steen & Gibbs 2004; Grosse *et al.* 2011). While each approach has merit for assessing road impacts to populations under practical sampling constraints, all are limited in their ability to estimate population consequences that are robust to important biases or assumptions (e.g., imperfect detection) (Langen *et al.* 2007; Crawford *et al.* 2014a). This is the first study to advance these approaches by employing multistate models, or any joint analysis (e.g., joint live-dead encounter model: Burnham 1993), for directly estimating demographic rates in the context of road mortality. We found empirical evidence that, when road threats were left unmitigated, mortality on the JIC significantly reduced per-capita survival of crossing females relative to non-crossers and caused severe population declines. Furthermore, estimated survival of crossing females is likely conservatively high since researchers intervened to capture live terrapins found on the road that may have been struck by vehicles. Our multistate model allowed for separate estimation of two processes that contribute to the cumulative impact of road mortality on local populations: the portion of the population exposed to the risk of road mortality and the magnitude of that risk (i.e., the degree to which survival is reduced when crossing). Disentangling these processes was essential during the projection phase of our study as we modeled the effects of two common road management actions designed to either prevent individuals from crossing the road (i.e., barriers) or increase survival of crossing individuals by targeting driver awareness (i.e., warning signage and education campaigns). Although obtaining multi-year mark-recapture datasets will be difficult in many contexts, our study demonstrates how these data can be used in integrated analyses to more realistically model the effects of roads to subsets of a population and measure population-level impacts needed to guide future research and management. Datasets requiring less effort (e.g., presence-absence data, population counts) can still be combined in integrated models to improve the precision of demographic estimates when mark-recapture data is (Schaub & Abadi 2011) or is not available (Zipkin *et al.* 2014).

Current management actions increased the probability of persistence, relative to the baseline scenario, but additional actions included in potential strategies were needed to further increase the probability of persistence and stabilize the terrapin population on the JIC. We detected several themes among strategies that yielded high (>80%) probabilities of persistence. Complementary actions targeting multiple threats, stage classes, and demographic rates were included in all top-performing strategies, while lower ranking strategies tended to have a singular focus. For example, strategy #7, which yielded the lowest probability of persistence other than strategy #1 (decreasing current efforts), included additional roadside barriers but no predator or vegetation management designed to increase fertility and recruitment rates. These results support previous conservation studies showing mitigation of multiple threats was required to ensure the viability of declining populations (Rhodes *et al.* 2011; Crawford *et al.* 2014a). Best strategies tended to include more extensive roadside barriers, which reduced the proportion of nesting terrapins crossing the road, as well as more intense predator management. It is not surprising that these strategies resulted in higher probabilities of persistence since PVA models were sensitive to changes in the probability of a female crossing. The probability of crossing was closely linked with adult female survival – the most sensitive model parameter – since non-crossing terrapins survived at a significantly higher rate than those exposed to vehicle mortality, even when we incorporated the largest effects of awareness actions. Thus, small changes in probability of crossing were accompanied by large changes in survival. Management strategies that prevent turtles from accessing the road should more efficiently increase population growth than strategies directly targeting survival of females on the road. Still, an important finding of our study was that flashing warning signage, the first-ever of its kind to target turtles, was associated with significant increases in survival of crossing individuals. Previous studies have found static warning signage to be ineffective (e.g., Putman 1997), presumably due to driver habituation to signage during periods when signs are present in the absence of animal crossing activity. Alternatively, we implemented signs to flash around concentrated peaks (3 hrs per day) of terrapin activity to avoid driver habituation. Strategically designing and implementing flashing signage may be a viable component of broader management strategies for diamondback terrapins and other species that demonstrate predictable and concentrated patterns of road-crossing activity.

Estimating parameters with integrated models requires several assumptions. Multiple datasets that are jointly analyzed must be independent of each other; although, simulation studies have found that violating this assumption has limited effects on parameter estimates (Abadi *et al.* 2010). This assumption holds for our study since we observed only three females (and no males) in both datasets, given low detection rates and total marked females at the local scale. A key assumption in joint analyses is that underlying demographic processes are shared and consistent.

We ensured consistency of female survival across datasets by parameterizing this rate with a grand mean that was informed by both datasets and incorporating additional terms to estimate survival of crossing and non-crossing females with road survey data. Site fidelity represented a terrapin being in the sampling area (i.e., in the creek and using the JIC to nest), which should be similar for the following reasons: 1) female terrapins complete temporary migrations from creeks to nesting areas prior to nesting (Tucker, Gibbons & Greene 2001; Sheridan *et al.* 2010), 2) Robust Design data was obtained from creeks adjacent to the JIC and sampled immediately prior to the nesting season each year, 3) the JIC is the largest and nearest nesting area within our population extent, 4) there is no evidence that terrapins forego nesting each year, and 5) we detected developing follicles in many female terrapins captured in creeks via palpation or radiographs taken at the GSTC (unpubl. data). Taken together, it is plausible that females encountered during creek sampling were staging there prior to nesting on the JIC. New females entering the population would exhibit the same movements patterns related to nesting, exposing them to sampling in both datasets in the same year. In the absence of other information, we can assume that annual entry into the population is proportional to those of local creeks. A related, implicit assumption is that the subpopulation sampled at the local (creek) scale was representative of all terrapins within the population (JIC) scale. A limitation of our study was that we could only sample two creeks, due to logistical constraints, to draw inferences at the local scale and inform inferences at the population scale. We have no evidence that sampled creeks differed from others within the population extent; marsh habitat characteristics were uniform across the population extent and we observed no additional threats occurring in sampled or nearby creeks. Although sampled creeks were close to the JIC, we can assume that any road-associated threats impact terrapins throughout the population extent equally because terrapins have been observed to travel over 1 km from tidal creeks to nest sites (Gibbons *et al.* 2001) and all creeks in the study area were located <400 m of the JIC (Crawford *et al.* 2014b). The extent to which survival, site fidelity, or entry differed between local and population scales (and datasets) would influence model convergence and uncertainty of shared parameters in the integrated model. Although we observed adequate convergence of models, sampling additional creeks in the future may reduce parameter uncertainty further. Despite being unable to test these assumptions at the present, we can rely on the robustness of our approach that combined datasets to reduce their individual potential biases and accounted for multiple sources of uncertainty to yield transparent predictions of population outcomes.

We made additional, simplifying assumptions in PVA models. We assumed no density dependence when projecting growth of the terrapin population. The population was declining rapidly under baseline and current conditions, indicating that it was below any carrying capacity and the influence of density dependence was minimal on model predictions. We assumed all changes to demographic rates via threats and management were additive and not compensatory. Although we did not test this assumption, demographic compensation has never been observed in turtle populations. To simplify the model, we used a mean fecundity rate for all adult females; although, many turtle species lay larger and more frequent clutches as females increase in age and size (Congdon & van Loben Sels 1993). Given that females that cross the road and survive likely repeat this behavior in subsequent years, road mortality may remove older females and reduce the mean fertility rate in road-impacted populations. Thus, we acknowledge literature-derived fertility rates used in this study could lead to overestimates of population growth and persistence. However, we expect population growth to be less sensitive to changes in fertility rates relative to effects of threats and management on other stages, especially adults.

Management implications

Given that the diamondback terrapin population on Jekyll Island is currently declining due to road-associated threats and potential strategies perform variably at increasing population growth, our findings give impetus for increased local management and have direct implications for which actions can most likely stabilize the population. Specifically, actions that prevent terrapins crossing at hot spots of activity complemented with predator and vegetation removal to increase recruitment will be required to maximize population persistence. For strategies that performed equally well at stabilizing the terrapin population, tradeoffs of other socioeconomic objectives (e.g., cost, driver safety, road aesthetics) need to be considered in decision making. We have also demonstrated that flashing warning signage can significantly reduce the likelihood of terrapin-vehicle collisions. Because signs were programmed to flash around daily high tide events that terrapins reliably use as a cue for nesting migrations (Feinberg & Burke 2003; Crawford *et al.* 2014b), these devices can be deployed in areas of frequent terrapin-vehicle collisions throughout the species range. Local managers should accompany implementation of management strategies with monitoring of their effects, especially for novel actions such as hybrid barriers and vegetation clearing. Monitoring data could inform estimates of management effects originally obtained from expert opinion, and site-specific PVAs should be updated iteratively as part of an adaptive management framework.

ACKNOWLEDGEMENTS

Funding to support this research was provided by the Daniel B. Warnell School of Forestry and Natural Resources and Graduate School, University of Georgia, and AGL Resources Foundation through the Jekyll Island Foundation through an assistantship to B.A.C. We thank GSTC staff, AmeriCorps members, and volunteers for their assistance throughout the project – especially M. Kaylor, D. Quinn, A. Grosse, R. Cozad, A. Gillis, L. Rodriguez , and S. Diltz. We thank associates at the Savannah River Ecology Lab and the Jekyll Island Authority for their continued collaborations. All methods were conducted in accordance with the recommendations for humane treatment of these animals for research and have been approved by the University of Georgia Institution Animal Care and Use Committee (Animal Use Protocol #: A2015 03-023-Y1-A0, expires May 20, 2018).

REFERENCES

Abadi, F., Gimenez, O., Arlettaz, R. & Schaub, M. (2010) An assessment of integrated population models: bias, accuracy, and violation of the assumption of independence. *Ecology,* **91,** 7-14.

Akçakaya, H.R. & Sjögren-Gulve, P. (2000) Population viability analyses in conservation planning: an overview. *Ecological Bulletins***,** 9-21.

Aresco, M.J. (2005a) The effect of sex-specific terrestrial movements and roads on the sex ratio of freshwater turtles. *Biological Conservation,* **123,** 37-44.

Aresco, M.J. (2005b) Mitigation measures to reduce highway mortality of turtles and other herpetofauna at a north Florida lake. *Journal of Wildlife Management,* **69,** 549-560.

Baird, R.C. (2009) Coastal urbanization: the challenge of management lag. *Management of Environmental Quality: An International Journal,* **20,** 371-382.

Brooks, S.P. & Gelman, A. (1998) General methods for monitoring convergence of iterative simulations. *Journal of computational and graphical statistics,* **7,** 434-455.

Buhlmann, K.A. & Osborn, C.P. (2011) Use of an artificial nesting mound by wood turtles (*Glyptemys insculpta*): a tool for turtle conservation. *Northeastern Naturalist,* **18,** 315-334.

Burnham, K. (1993) A theory for combined analysis of ring recovery and recapture data. *Marked individuals in the study of bird population.* (eds J. Lebreton & P. North), pp. 199-213. Birkhauser Verlag, Basel, Switzerland.

Butler, J., Heinrich, G. & Seigel, R. (2006) Third workshop on the ecology, status, and conservation of diamondback terrapins (*Malaclemys terrapin*): results and recommendations. *Chelonian Conservation and Biology,* **5,** 331-334.

Caswell, H. (1989) *Matrix population models: construction, analysis, and interpretation*. Sinauer Associates, Sunderland, U.S.A.

Chambers, R.M. & Maerz, J.C. (*in press*) Terrapin bycatch in the blue crab fishery. *Ecology and conservation of the diamondback terrapin* (eds W.M. Roosenburg & V.S. Kennedy). Johns Hopkins University Press, Baltimore, MD.

Clark, J.S. & Bjørnstad, O.N. (2004) Population time series: process variability, observation errors, missing values, lags, and hidden states. *Ecology,* **85,** 3140-3150.

Congdon, J.D., Dunham, A.E. & van Loben Sels, R.C. (1993) Delayed sexual maturity and demographics of Blanding's turtles (*Emydoidea blandingii*) - implications for conservation and management of long-lived organisms. *Conservation Biology,* **7,** 826-833.

Congdon, J.D. & van Loben Sels, R.C. (1993) Relationships of reproductive traits and body size with attainment of sexual maturity and age in Blanding's turtles (*Emydoidea blandingi*). *Journal of Evolutionary Biology,* **6,** 547-557.

Crawford, B.A. (2015) Lessons from terrapin mortality and management on the Jekyll Island Causeway, Georgia, USA. *Roads and ecological infrastructure: Concepts and applications for small animals,* pp. 140-145. Johns Hopkins University Press, Baltimore, MD.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A. & Norton, T.M. (2014a) Estimating the consequences of multiple threats and management strategies for semi-aquatic turtles. *Journal of Applied Ecology,* **51,** 359-366.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A., Norton, T.M. & Albeke, S.E. (2014b) Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology,* **51,** 367-375.

Dorcas, M.E., Willson, J.D. & Gibbons, J.W. (2007) Crab trapping causes population decline and demographic changes in diamondback terrapins over two decades. *Biological Conservation,* **137,** 334-340.

Ewert, M.A., Jackson, D.R. & Nelson, C.E. (1994) Patterns of temperature dependent sex determination in turtles. *Journal of Experimental Zoology,* **270,** 3-15.

Feinberg, J.A. & Burke, R.L. (2003) Nesting ecology and predation of diamondback terrapins, *Malaclemys terrapin*, at Gateway National Recreation Area, New York. *Journal of Herpetology,* **37,** 517-526.

Game, E.T., Meijaard, E., Sheil, D. & McDonald-Madden, E. (2014) Conservation in a wicked complex world: Challenges and solutions. *Conservation Letters,* **7,** 271-277.

Georgia Department of Transportation (2014) Georgia's State Traffic and Report Statistics (STARS). Available from http://www.dot.state.ga.us/statistics/stars/Pages/GlynnTraffic.aspx (accessed on 19 Oct 2011).

Gibbons, J.W., Lovich, J.E., Tucker, A.D., Fitzsimmons, N.N. & Greene, J.L. (2001) Demographic and ecological factors affecting conservation and management of the Diamondback Terrapin (*Malaclemys terrapin*) in South Carolina. *Chelonian Conservation and Biology,* **4,** 66-74.

Gibbs, J.P. & Shriver, W.G. (2002) Estimating the effects of road mortality on turtle populations. *Conservation Biology,* **16,** 1647-1652.

Gilliand, S.C., Chambers, R.M. & LaMar, M.D. (2014) Modeling the effects of crab potting and road traffic on a population of diamondback terrapins. *Proceedings of the Symposium on BEER*.

Grosse, A.M., Crawford, B.A., Maerz, J.C., Buhlmann, K.A., Norton, T.M., Kaylor, M. & Tuberville, T.D. (2015) Effects of Vegetation Structure and Artificial Nesting Habitats on Hatchling Sex Determination and Nest Survival of Diamondback Terrapins. *Journal of Fish and Wildlife Management,* **6,** 19-28.

Grosse, A.M., Maerz, J.C., Hepinstall-Cymerman, J. & Dorcas, M.E. (2011) Effects of roads and crabbing pressures on diamondback terrapin populations in coastal Georgia. *Journal of Wildlife Management,* **75,** 762-770.

Hart, K.M. (2005) *Population biology of diamondback terrapins (Malaclemys terrapin): defining and reducing threats across their geographic range*. PhD thesis, Duke University.

Hels, T. & Buchwald, E. (2001) The effect of road kills on amphibian populations. *Biological Conservation,* **99,** 331-340.

Heppell, S.S., Crowder, L.B. & Crouse, D.T. (1996) Models to evaluate headstarting as a management tool for long-lived turtles. *Ecological Applications,* **6,** 556-565.

Isdell, R.E., Chambers, R.M., Bilkovic, D.M. & Leu, M. (2015) Effects of terrestrial–aquatic connectivity on an estuarine turtle. *Diversity and Distributions,* **21,** 643-653.

Kéry, M. & Schaub, M. (2012) *Bayesian population analysis using WinBUGS: A hierarchical perspective*. Academic Press, Oxford, UK.

Krueger, T., Page, T., Hubacek, K., Smith, L. & Hiscock, K. (2012) The role of expert opinion in environmental modelling. *Environmental Modelling & Software,* **36,** 4-18.

Langen, T.A., Machniak, A., Crowe, E.K., Mangan, C., Marker, D.F., Liddle, N. & Roden, B. (2007) Methodologies for surveying herpetofauna mortality on rural highways. *Journal of Wildlife Management*, **71,** 1361-1368.

Lefkovitch, L. (1965) The study of population growth in organisms grouped by stages. *Biometrics*, **21,** 1-18.

Link, W.A. & Eaton, M.J. (2012) On thinning of chains in MCMC. *Methods in Ecology and Evolution*, **3,** 112-115.

Lovich, J.E. & Gibbons, J.W. (1990) Age at maturity influences adult sex-ratio in the turtle *Malaclemys terrapin*. *Oikos*, **59,** 126-134.

Maerz, J.C., Seigel, R.A. & Crawford, B.A. (*in press*) Terrapin conservation: Mitigating habitat loss, road mortality, and subsidized predators. *Ecology and conservation of the diamondback terrapin* (eds W.M. Roosenburg & V.S. Kennedy). Johns Hopkins University Press, Baltimore, MD.

McGowan, C.P., Runge, M.C. & Larson, M.A. (2011) Incorporating parametric uncertainty into population viability analysis models. *Biological Conservation*, **144,** 1400-1408.

Mitro, M.G. (2003) Demography and viability analyses of a diamondback terrapin population. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, **81,** 716-726.

Moore, C.T., Converse, S.J., Folk, M.J., Runge, M.C. & Nesbitt, S.A. (2012) Evaluating release alternatives for a long-lived bird species under uncertainty about long-term demographic rates. *Journal of Ornithology*, **152,** 339-353.

Morris, W.F. & Doak, D.F. (2002) *Quantitative conservation biology: theory and practice of population viability analysis*. Sinauer Associates, Sunderland, Massachusetts.

Mumme, R.L., Schoech, S.J., Woolfenden, G.E. & Fitzpatrick, J.W. (2000) Life and death in the fast lane: demographic consequences of road mortality in the Florida Scrub-Jay. *Conservation Biology,* **14,** 501-512.

Munscher, E.C., Kuhns, E.H., Cox, C.A. & Butler, J.A. (2012) Decreased nest mortality for the Carolina diamondback terrapin (*Malaclemys terrapin centrata*) following removal of raccoons (*Procyon lotor*) from a nesting beach in northeastern Florida. *Herpetological Conservation and Biology,* **7,** 176-184.

Pollock, K.H. (1982) A capture-recapture design robust to unequal probability of capture. *The Journal of Wildlife Management***,** 752-757.

Putman, R.J. (1997) Deer and road traffic accidents: Options for management. *Journal of Environmental Management,* **51,** 43-57.

Quinn, D.P., Kaylor, M., Norton, T.M. & Buhlmann, K.A. (2015) Nesting mounds with protective boxes and an electric wire as tools to mitigate diamond-backed terrapin (*Malaclemys terrapin*) nest predation. *Herpetological Conservation and Biology,* **10,** 969-977.

R Core Team (2013) R: A language and environment for statistical computing. *R Foundation for Statistical Computing*. Vienna, Austria. Available from http://www.R-project.org/

Rhodes, J.R., Ng, C.F., de Villiers, D.L., Preece, H.J., McAlpine, C.A. & Possingham, H.P. (2011) Using integrated population modelling to quantify the implications of multiple threatening processes for a rapidly declining population. *Biological Conservation,* **144,** 1081-1088.

Roosenburg, W.M. (1991) The diamondback terrapin: Population dynamics, habitat requirements, and opportunities for conservation. *New Perspectives in the Chesapeake*

System: A Research and Management Partnership. Proceedings of a Conference. CRC Publ, pp. 227-234.

Roosenburg, W.M., Cresko, W., Modesitte, M. & Robbins, M.B. (1997) Diamondback terrapin (*Malaclemys terrapin*) mortality in crab pots. *Conservation Biology,* **11,** 1166-1172.

Row, J.R., Blouin-Demers, G. & Weatherhead, P.J. (2007) Demographic effects of road mortality in black ratsnakes (*Elaphe obsoleta*). *Biological Conservation,* **137,** 117-124.

Royle, J.A. & Dorazio, R.M. (2012) Parameter-expanded data augmentation for Bayesian analysis of capture–recapture models. *Journal of Ornithology,* **152,** 521-537.

Schaub, M. & Abadi, F. (2011) Integrated population models: a novel analysis framework for deeper insights into population dynamics. *Journal of Ornithology,* **152,** 227-237.

Schmolke, A., Thorbek, P., DeAngelis, D.L. & Grimm, V. (2010) Ecological models supporting environmental decision making: a strategy for the future. *Trends in Ecology & Evolution,* **25,** 479-486.

Sheridan, C.M., Spotila, J.R., Bien, W.F. & Avery, H.W. (2010) Sex-biased dispersal and natal philopatry in the diamondback terrapin, Malaclemys terrapin. *Molecular Ecology,* **19,** 5497-5510.

Spiegelhalter, D.J., Best, N.G., Carlin, B.P. & Van Der Linde, A. (2002) Bayesian measures of model complexity and fit. *Journal of the Royal Statistical Society: Series B (Statistical Methodology),* **64,** 583-639.

Steen, D.A. & Gibbs, J.P. (2004) Effects of roads on the structure of freshwater turtle populations. *Conservation Biology,* **18,** 1143-1148.

Su, Y.-S. & Yajima, M. (2012) R2jags: A Package for Running jags from R. Available from R package version 0.03-08, URL http://CRAN.R-project.org/package=R2jags.

Szerlag-Egger, S. & McRobert, S.P. (2007) Northern diamondback terrapin occurrence, movement, and nesting activity along a salt marsh access road. *Chelonian Conservation and Biology*, **6,** 295-301.

Tempel, D.J., Peery, M. & Gutiérrez, R. (2014) Using integrated population models to improve conservation monitoring: California spotted owls as a case study. *Ecological Modelling,* **289,** 86-95.

Tucker, A.D., Gibbons, J.W. & Greene, J.L. (2001) Estimates of adult survival and migration for diamondback terrapins: conservation insight from local extirpation within a metapopulation. *Canadian Journal of Zoology,* **79,** 2199-2209.

Wade, P.R. (2002) Bayesian population viability analysis. *Population viability analysis***,** 213-238.

Weisrock, D. & Janzen, F. (1999) Thermal and fitness-related consequences of nest location in painted turtles (Chrysemys picta). *Functional Ecology,* **13,** 94-101.

Zipkin, E.F., Thorson, J.T., See, K., Lynch, H.J., Grant, E.H.C., Kanno, Y., Chandler, R.B., Letcher, B.H. & Royle, J.A. (2014) Modeling structured population dynamics using data from unmarked individuals. *Ecology*, **95,** 22-29.

Table 4.1. Parameter estimates used in a population viability analysis for diamondback terrapins.

Parameter	**Value**	**SD**	**Description**	**Source**
ϕ_{Xh} , ϕ_{Yh}	0.530	-	Survival of hatchlings (males and females)	Gilliand et al. 2013
ϕ_{Xj} , ϕ_{Yj}	0.570	-	Survival of juveniles (males and females)	Mitro 2003
ϕ_{Ya}	0.680	0.075	Survival of adult males	Estimated
ϕ_{Xa}	0.720	0.082	Survival of adult females (mean)	Estimated
ϕ_c (before)	0.235	0.069	Survival of crossing females before warning signage	Estimated
ϕ_c (after)	0.532	0.139	Survival of crossing females after warning signage	Estimated
ϕ_{nc}	0.903	0.061	Survival of adult females not crossing roads	Estimated
γ	0.683	0.138	Probability of adult females using causeway for nesting	Estimated
c	0.531	0.033	Probability of adult females crossing the road	Estimated
b_{Ya}	0.263	0.164	Probability of males entering the population	Estimated
b_{Xa}	0.187	0.070	Probability of females entering the population	Estimated
k	6.9	1.5	Mean clutch size	Zimmerman 1992
u	2	-	Mean clutch frequency per year	Godwin 1994
ϕ_e	0.790	0.065	Natural survival of eggs	Grosse et al. 2015
ns_{hab}			Nest success by habitat type - the proportion of nests not depredated	
open	0.548	0.063	Open	Grosse et al. 2015
hedges	0.156	0.054	Hedges	Grosse et al. 2015
mounds	0.963	0.026	Mounds with electrified boxes	D. Quinn & GSTC, unpubl. data
h_{hab}		LCI	Proportion of female hatchlings by habitat type	
open	1	(0.51)	Open	Grosse et al. 2015
hedges	0.15	(0.00)	Hedges	Grosse et al. 2015
mounds	1	(0.51)	Mounds with electrified boxes	Grosse et al. 2015
A_Y	5		Age of first reproduction (males)	Lovich and Gibbons, 1990
A_X	6		Age of first reproduction (females)	Lovich and Gibbons, 1990

Table 4.2. Sensitivity of mean population growth rate (λ) to proportional changes in demographic rates in the baseline population viability model. Values represent differences in resulting λ from the baseline. The bottom three rows indicate mean λ when sex-specific juvenile graduation probabilities were replaced with sex-specific entry probabilities estimated from the integrated model and when habitat-specific sex ratios were replaced with lower proportions of females.

	Mean λ	
Parameter	**-5%**	**+5%**
Baseline λ	0.796	
ϕ_e	-0.007	0.006
ϕ_{Xh} , ϕ_{Yh}	-0.008	0.000
ϕ_{Yj}	-0.004	0.000
ϕ_{Xj}	-0.015	0.022
ϕ_{Ya}	-0.002	0.001
$\phi_{Xa.c}$	-0.009	0.009
$\phi_{Xa.nc}$	-0.023	0.018
γ	-0.000	0.002
c	-0.017	0.000
bmale	0.789	
bfemale	0.920	
h_{hab}	0.762	

Table 4.3. Sensitivity of population persistence to changes in experts' estimates of management effects on demographic rates across management strategies. Values indicate differences in resulting probabilities of persistence from the mean when lower (LCI) and upper (UCI) estimates of effects are used in the population viability model. Dashes indicate strategies without management actions that affected demographic rates.

Demographic rate		**Survival of crossers**		**Proportion blocked**		**Nest success**	
Strategy	**Mean**	**LCI**	**UCI**	**LCI**	**UCI**	**LCI**	**UCI**
Baseline	0.015	-	-	-	-	-0.008	0.016
Current	0.369	-	-	-0.009	0.008	-0.093	0.077
1	0.377	-	-	-	-	-0.096	0.060
2	0.964	-	-	-0.011	0.008	-0.008	0.001
3	0.814	-	-	-0.052	0.041	-0.077	0.039
4	0.833	-	-	-0.032	0.018	-0.067	0.020
5	0.754	-0.010	0.055	-0.065	0.088	-0.079	0.055
6	0.401	-0.023	0.084	0.020	0.013	-0.081	0.084
7	0.813	-0.027	0.021	-0.016	0.015	-0.050	0.021
8	0.380	-	-	0.002	0.003	-0.098	0.063
9	0.668	-0.040	0.057	-0.019	0.020	-0.099	0.057
10	0.520	-0.037	0.063	-0.010	0.031	-0.104	0.063
11	0.425	-0.035	0.088	-0.002	0.012	-0.097	0.088
12	0.661	-	-	-0.013	0.016	-0.101	0.055
13	0.652	-0.039	0.042	-0.014	0.005	-0.076	0.042
14	0.526	-	-	-0.029	0.043	-0.108	0.099
15	0.836	-0.009	0.031	-0.027	0.031	-0.063	0.031
16	0.823	-0.013	0.038	-0.041	0.046	-0.054	0.038
17	0.551	-	-	-0.040	0.101	-0.113	0.093
18	0.820	-	-	-0.041	0.042	-0.061	0.048
19	0.902	-	-	-0.045	0.027	-0.039	0.016

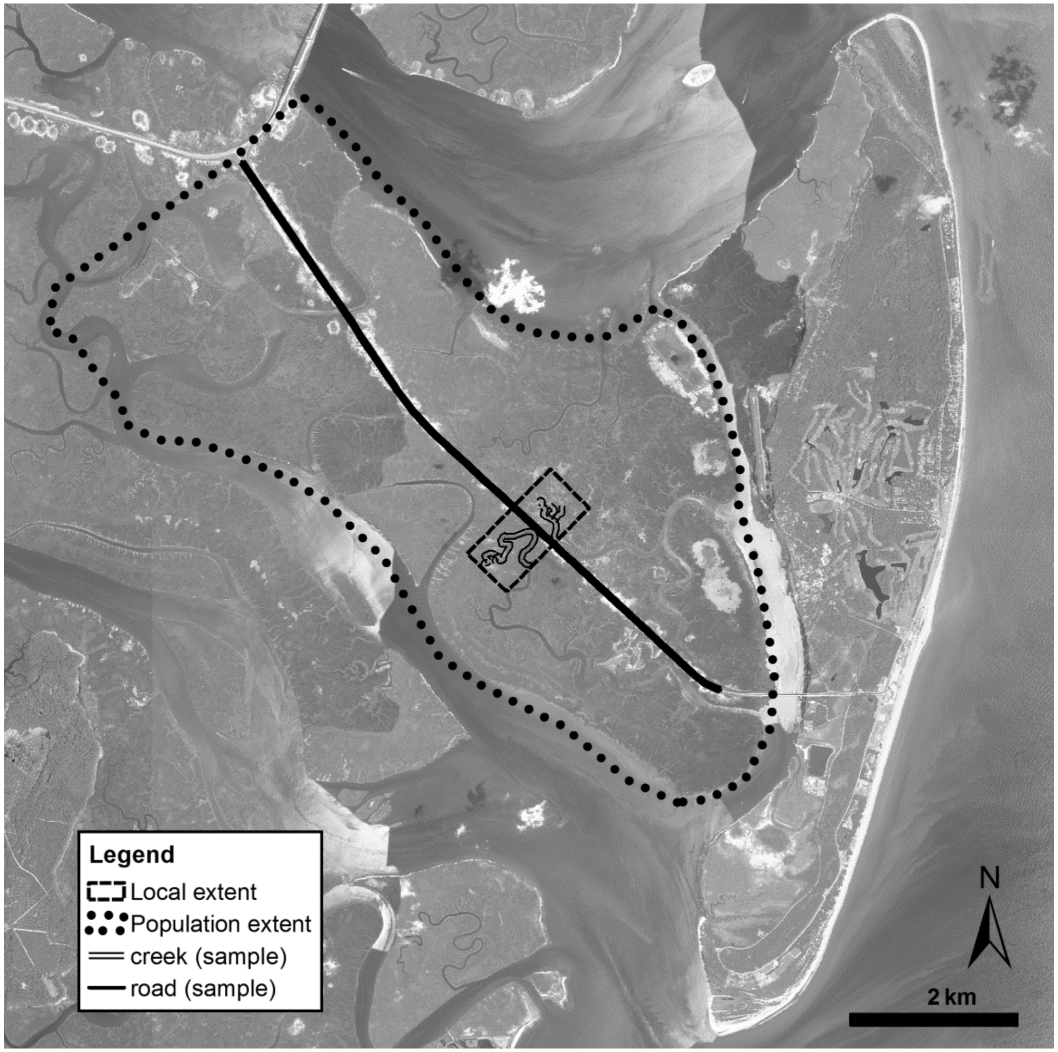


Figure 4.1. Map of areas sampled for each data set used to estimate parameters of the diamondback terrapin population of interest inhabiting the salt marsh peninsula around the Jekyll Island Causeway, Jekyll Island, Georgia, USA.

Figure 4.2. Management actions for diamondback terrapins deployed on the Jekyll Island Causeway, Jekyll Island, Georgia, USA. The hybrid barrier (a) was placed on elevated mounds and composed of nest boxes with openings towards the marsh allowing nesting terrapins to enter. Two flashing warning signs (b) were programmed to flash around the daily, diurnal high tide, corresponding with the temporal peak of nesting activity.

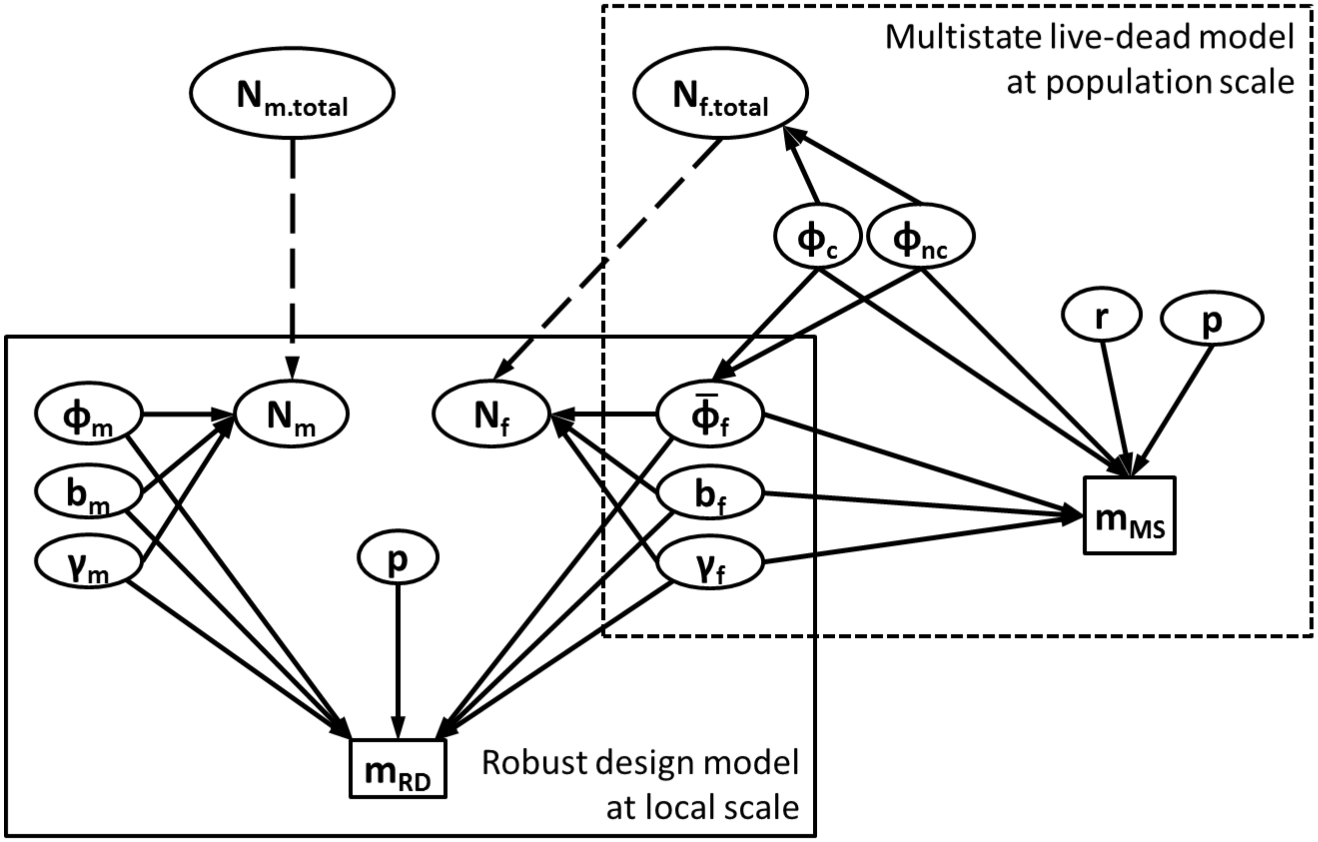


Figure 4.3. Directed acyclic graph of the integrated model, showing demographic parameters that were independently or jointly estimated from separate capture-mark-recapture datasets. Robust Design CMR data obtained through sampling of two creeks adjacent to the Jekyll Island Causeway was used to estimate male and female parameters at a local scale; live-dead capture-mark-recapture/recover data obtained through road surveying across the Jekyll Island Causeway was used to estimate female parameters and derive male abundance at the population scale. Estimated and derived parameters are represented by ovals and the data are represented

by rectangles. Solid arrows represent dependencies between nodes that were implicit in the model; dashed arrows represent assumed dependences between local and population-level abundances that were used in the population viability analysis. Node notations: m capture–recapture data; φ survival; b entry; γ site fidelity; p capture; r recovery; N population size. Note that the priors are not shown on this graph. Subscript notations: m male; f female; c crossers; nc non-crossers; total population scale across the Jekyll Island Causeway.

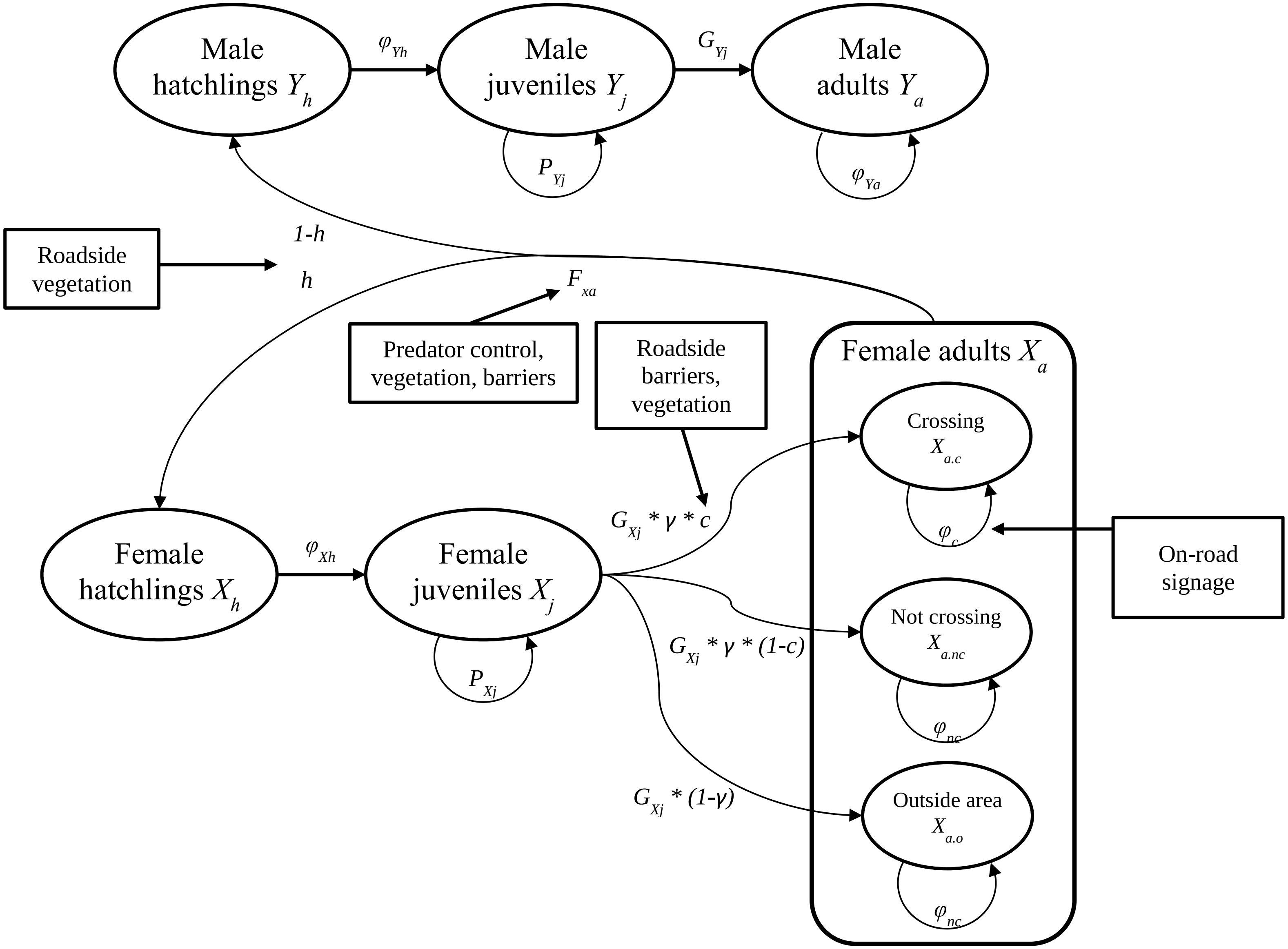

Male hatchlings Y_h
φ_{Yh}
Male juveniles Y_j
G_{Yj}
Male adults Y_a
P_{Yj}
φ_{Ya}
Roadside vegetation
$1\text{-}h$
h
F_{xa}
Predator control, vegetation, barriers
Roadside barriers, vegetation
Female adults X_a
Crossing $X_{a.c}$
φ_c
On-road signage
$G_{Xj} * \gamma * c$
Female hatchlings X_h
φ_{Xh}
Female juveniles X_j
P_{Xj}
Not crossing $X_{a.nc}$
$G_{Xj} * \gamma * (1\text{-}c)$
φ_{nc}
$G_{Xj} * (1\text{-}\gamma)$
Outside area $X_{a.o}$
φ_{nc}

Figure 4.4. Male and female three-stage model for projecting dynamics of a road-impacted diamondback terrapin population (adapted from Gilland *et al.* 2013). Ovals represent life stages and states; rectangles represent management strategies expected to impact model parameters; symbols along arrows represent parameters described in Table 4.1. Transition rates between adult female states are not represented with arrows, but are functions of γ and c described in Table 4.1.

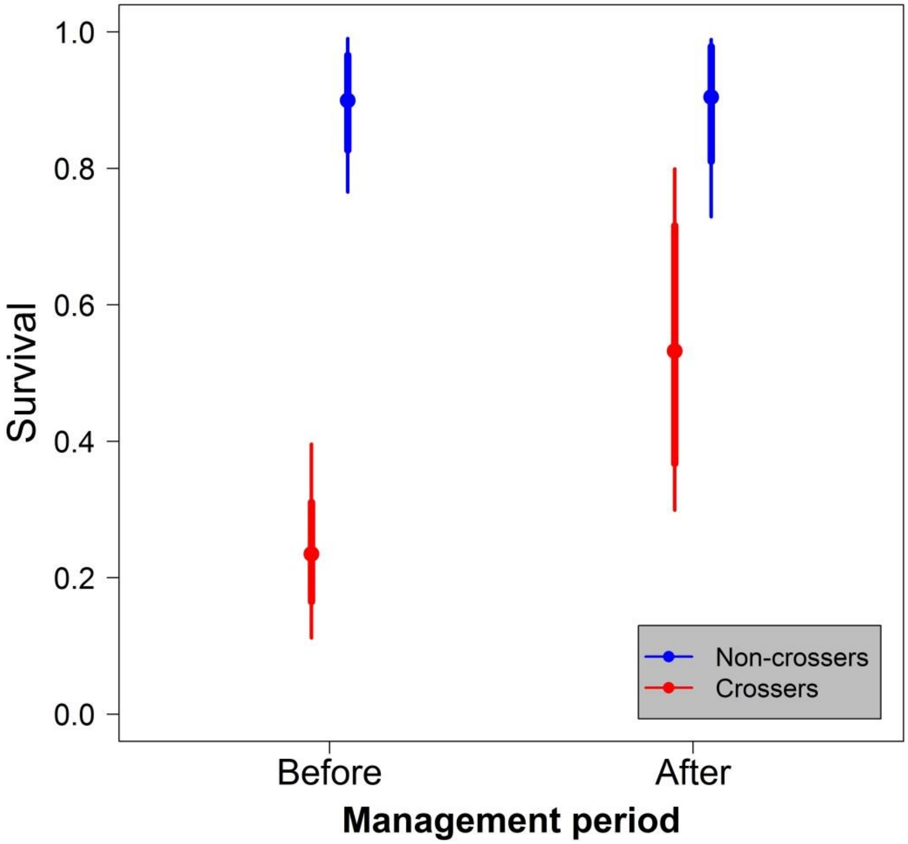


Figure 4.5. Effects of road crossing status and pre-post management period on mean (± 95% BCI) annual survival for adult female diamondback terrapins on Jekyll Island, GA.

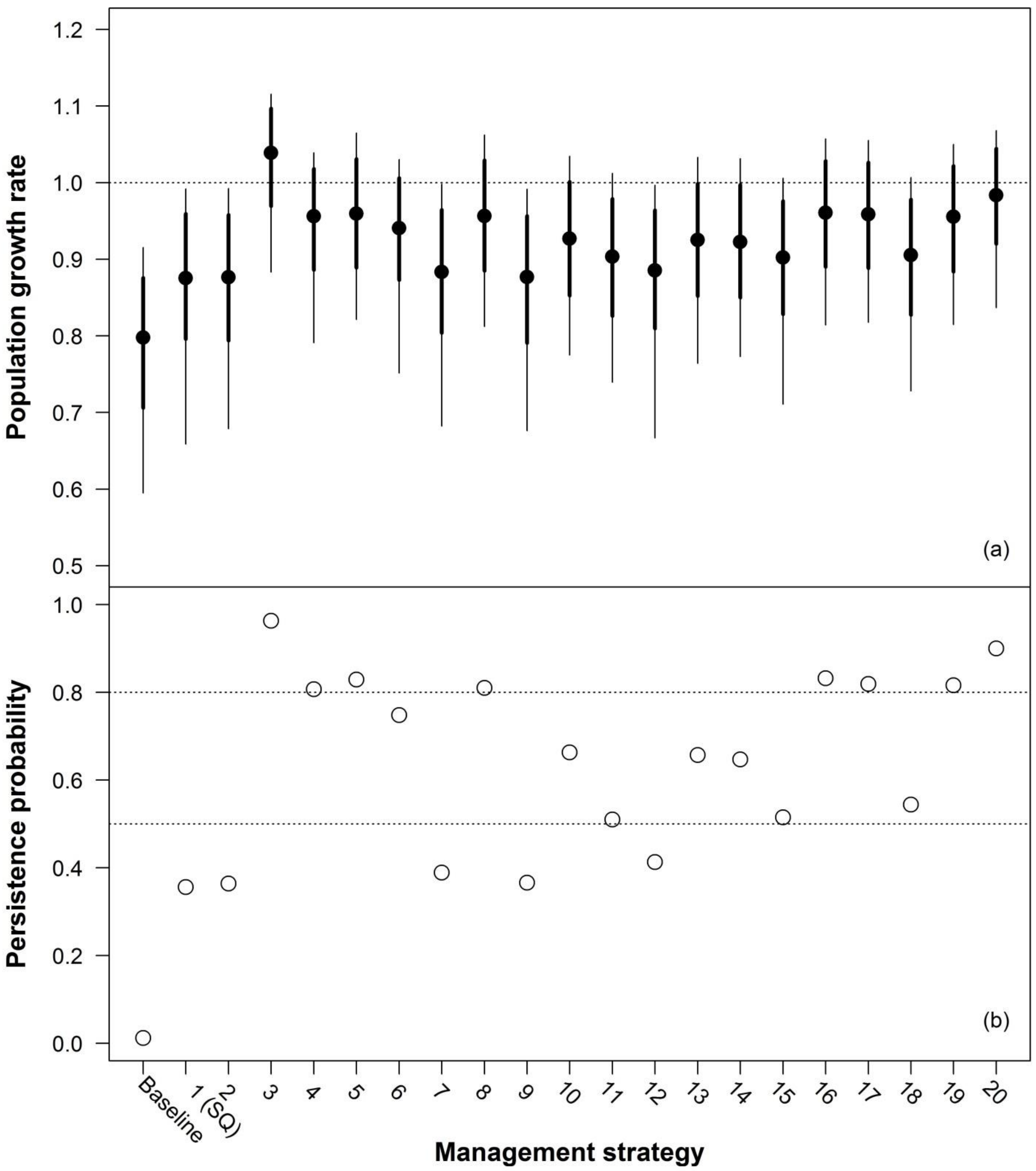


Figure 4.6. Simulation outcomes (a: population growth rate λ; b: persistence probability) for the diamondback terrapin population on Jekyll Island, GA under baseline conditions and 20 potential management strategies, including keeping the status quo (SQ) of current management.

CHAPTER 5

ROAD MANAGEMENT FOR A DECLINING SPECIES: INTEGRATING MULTIPLE OBJECTIVES THROUGH STRUCTURED DECISION MAKING[3]

---

[3] Crawford, B.A., J.C. Maerz, N. Heynen, T.M. Norton, and C.T. Moore, To be submitted to *Conservation Biology*

ABSTRACT

Natural resource managers frequently face difficult decisions that attempt to promote often-competing objectives (biodiversity and human well-being) in the presence of diverse stakeholder preferences and uncertainty of decision outcomes. Structured decision making (SDM) has emerged as an effective tool for addressing such contexts, but SDM has not been applied to one pervasive and challenging conservation issue: roads. Roads present multiple threats to nearby wildlife populations at various scales; however, management attempting to reduce impacts to wildlife must also consider roads as valuable infrastructure important to people's daily lives and local economies. We describe the co-development of a multi-objective SDM framework with local stakeholders to identify management decisions using the diamondback terrapin (*Malaclemys terrapin*) as a focal species and the Jekyll Island Causeway (Jekyll Island, Georgia, USA) as a model system. Stakeholders identified five fundamental management objectives: maximizing terrapin persistence, human safety, patron satisfaction, transferable knowledge (learning), and minimizing net cost. We used ecological and human dimensions data, as well as expert opinion, to model the effects of 20 alternative management strategies on objective outcomes. Decision analysis showed that i) stakeholders varied in their preferences for objectives, but terrapin persistence and human safety were most important, on average, ii) the best-performing strategy overall did not rank best for any one objective but included complementary actions (roadside barriers, warning signage, conservation awareness campaigns) that allowed it to perform well across most objectives, and iii) the best strategy was insensitive to stakeholder-specific preferences. Our results provide direct recommendations for addressing terrapin conservation and socioeconomic goals for Jekyll Island. Furthermore, our study demonstrates the efficacy of applying SDM to road management contexts and the benefit

of using human dimensions data to more accurately measure preferences of large stakeholder groups and model decision-outcomes.

INTRODUCTION

There is a growing awareness that most problems facing the field of conservation are wicked in nature (Game *et al.* 2014). Conservation issues often involve multiple objectives that conflict or compete for limited resources (e.g., protect biodiversity, promote economic development, minimize cost), such that decision makers must assess the tradeoffs between meeting one objective or another (Hirsch *et al.* 2011; Converse *et al.* 2013). Reaching decisions is made more challenging by substantial uncertainty about the current state of the system being considered and its response to potential management alternatives (Hirsch *et al.* 2011; McShane *et al.* 2011). In light of these challenges, decision making approaches are needed to help managers navigate complex and uncertain conservation issues and produce robust, equitable outcomes.

Structured decision making (SDM), a form of decision analysis, has emerged in recent decades as a systematic process in which stakeholders engage to address complex conservation issues (Gregory *et al.* 2012). SDM identifies management alternatives that optimally meet objectives through a process designed to clearly separate decision components that are based on values with those based on science (Clemen & Reilly 2013; Conroy & Peterson 2013). Stakeholders collaboratively advance through the steps of SDM by framing the problem, defining relevant objectives, developing management alternatives, predicting consequences to the system, and examining tradeoffs to identify optimal decisions (Gregory *et al.* 2012). Recognized strengths of SDM include the following: i) transparency throughout the process that allows a clearer understanding of uncertainty and anticipated tradeoffs between outcomes of competing objectives, ii) inclusivity of stakeholders' knowledges and perspectives at each step, and iii) emphasis on defining objectives before prioritizing management options (Conroy & Peterson 2013). We refer readers to Clemen and Reilly (2013) for a deeper coverage of SDM, in general, and to Conroy and Peterson (2013) for SDM within the context of natural resource management. SDM has been effectively applied to many wildlife management contexts that focus primarily on meeting biological (population) objectives and linking these to actions via rigorous ecological models (e.g., Blomquist *et al.* 2010; Converse *et al.* 2013). Social objectives (e.g., maximize public satisfaction, agency public relations) are less frequently included in decision contexts and, if included, are evaluated using coarse metrics and expert opinion (e.g., Converse *et al.* 2013). However, it is the norm, and not the exception, that conservation issues include social dimensions that will be affected by decisions (Hirsch *et al.* 2011). Therefore, SDM may produce more robust decisions by incorporating data (e.g., the public's management preferences, attitudes) and approaches from the fields of human dimensions and social sciences that directly inform social objectives. Even with its current gaps, SDM has yielded defendable decisions across many conservation contexts, including endangered species management (e.g., Gregory & Long 2009), invasive species management (e.g., Liu *et al.* 2012), reintroduction (e.g., Converse *et al.* 2013), and formation of policy regulating species harvest (e.g., Nichols & Williams 2006). However, limited work has applied formal decision making approaches to another widespread and wicked conservation problem: roads.

Roads present the field of conservation management with many challenging elements ideally suited for SDM. Multiple management objectives arise as roads simultaneously represent valuable infrastructure important to people's daily lives and a pervasive threat to wildlife (Fahrig & Rytwinski 2009; Andrews, Nanjappa & Riley 2015). Strategies designed to benefit wildlife may impinge upon other social or economic objectives. For example, management actions, such as reducing speed limits, constructing barriers to keep animals from entering roadways, or restricting vehicle access, likely have direct or unforeseen consequences to drivers' happiness or safety that will need to be traded off with wildlife outcomes. Evaluating the acceptability of these tradeoffs will depend on the values and preferences of public user groups (Stout *et al.* 1993; Crawford, Poudyal & Maerz 2015). Complexity and uncertainty surrounding road-wildlife interactions make decisions more difficult. In order to sustain wildlife populations, management must consider multiple road-associated threats that can concurrently contribute to population declines (Rhodes *et al.* 2011; Crawford *et al.* 2014a). These include direct collisions with vehicles, reduction and fragmentation of wildlife habitat, impediment to animal movement, predation by species subsidized by human activities (e.g., raccoons, *Procyon lotor*), and alteration of species behavior (Trombulak & Frissell 2000; Fahrig & Rytwinski 2009). Numerous management actions exist to mitigate each threat (Glista, DeVault & DeWoody 2009). Although many have successfully reduced threats (e.g., roadside barriers for wildlife: Aresco 2005), others have yielded limited or uncertain conservation outcomes (e.g., warning signage for drivers: Putman 1997; Sullivan *et al.* 2004). Designing and implementing cost-effective actions is difficult since road threats to local species are often spatially or temporally diffuse; however, concentrated patterns of threats (“hot spots” and “hot moments”) can be identified as targets for management in some contexts (Beaudry, Demaynadier & Hunter 2010; Cureton & Deaton 2012; Crawford *et al.* 2014b).

We describe the co-development of an SDM framework with local stakeholders to identify conservation management decisions using the diamondback terrapin (*Malaclemys terrapin*) as a focal species and the Jekyll Island Causeway (Jekyll Island, Georgia, USA) as a model system. Representative of many socio-ecological systems, the Jekyll Island Causeway provides a context for addressing, through decision analysis, i) multiple threats to the terrapin, a species of conservation concern, ii) shared and conflicting objectives of multiple stakeholders, and iii) uncertainty of expected consequences for management alternatives. Our goal was for the Jekyll Island State Park Authority, the agency commissioned with managing the area, to use the framework to implement strategies on the Causeway that address terrapin conservation and socioeconomic objectives. This work builds on previous research where we identified hot spots and hot moments of wildlife-vehicle collisions to inform management targets (Crawford *et al.* 2014b), measured road user groups' preferences for management actions (Crawford, Poudyal & Maerz 2015), and estimated population outcomes given no management, current conditions, and potential management strategies (Chapter 4).

STUDY AREA AND FOCAL SPECIES

The 8.7-km Downing-Musgrove Causeway (aka Jekyll Island Causeway: JIC) is the only road connecting the mainland with Jekyll Island, GA, USA (31.08°N, 81.47°W). The JIC is characterized as a high-speed (89 km/hr [55 mph]) state highway with average annual daily traffic (AADT) of 3,440 vehicles/day that peaks from May through July, corresponding with increased summer tourism (Georgia Department of Transportation 2014). Roadside habitat varies from open areas to densely vegetated hedgerows of cedars (*Juniperus virginiana*) and wax myrtles (*Myrica cerifera*) maintained as a wind break. The JIC bisects a peninsula of salt marsh roughly 32 km^2 in area consisting of a network of intertidal creeks and high marsh dominated by *Spartina spp*.

Diamondback terrapins inhabit salt marshes along the Eastern and Gulf Coasts of the United States – regions experiencing the fastest annual increases in both the densities and traffic loads of roads (Baird 2009). Each summer, adult females complete overland movements in search of open, elevated nesting habitat – the same habitat provided by roadsides in human-developed landscapes (Butler, Heinrich & Seigel 2006; Szerlag-Egger & McRobert 2007).

Representative of many high-traffic coastal areas, the JIC is a regional hot spot of mortality where 100-400 adult female terrapins are struck and killed each year while attempting to cross the road to nest (recorded since 2007: Crawford *et al.* 2014b). Populations are also impacted by indirect, road-associated threats, such as subsidized predators (e.g., *Procyon lotor*) that can cause high rates (50%-90%) of egg mortality near roads throughout the species range (Feinberg & Burke 2003; Szerlag & McRobert 2006). Roadside habitat used for nesting influences nest predation rates as well as hatchling sex ratios, where nests laid in warmer, open areas are more likely to survive and produce females due to environmental sex determination (Grosse *et al.* 2015). Given these threats, the terrapin population surrounding the JIC is predicted to decline in the absence of management (Chapter 4; Crawford *et al.* 2014a).

METHODS

Stakeholders and workshop series

We co-developed the SDM framework with a team of fourteen representatives across eight stakeholder groups who had the ability to affect or be affected by management decisions for the JIC. Within the decision context, six distinct groups were full stakeholders and two organizations were partial stakeholders (i.e., they had departmental missions linked to some but not all of the fundamental objectives in our decision context [see below]). The decision maker was the Board of Directors of the Jekyll Island State Park Authority (JIA: http://www.jekyllisland.com/jekyll-island-authority/), represented by its Executive Director. JIA was created and commissioned by the Georgia state government with overseeing the development of Jekyll Island while protecting its natural resources. The JIA's Georgia Sea Turtle Center (GSTC: http://www.georgiaseaturtlecenter.org/) was established in 2007 to carry out and promote conservation efforts on Jekyll Island through rehabilitation, research, and outreach and education. We considered the GSTC and JIA as separate stakeholders since the former's primary goal is conservation while the latter must consider conservation and development goals. Other represented stakeholders included Jekyll Island patron groups using the causeway: visitors, residents, and employees. The Golden Isles Convention and Visitors Bureau represented an estimated 1.6 million visitors who annually travel to Jekyll Island and encompass the largest patron group (E. Garvey [JIA]; unpubl. data). An estimated 805 people reside on Jekyll Island (U.S. Census Bureau 2010) who were represented by the Jekyll Island Citizens Association (JICA: http://www.jekyllcitizens.org/). The Jekyll Island Business Association represented roughly 150 employees associated with JIA and non-JIA businesses (island hotels, restaurants, golf courses, and attractions). Multiple conservation non-governmental organizations operate on the Georgia coast and actively work toward ensuring responsible management and development, and this stakeholder group was represented by the advocacy group One Hundred Miles based locally in Brunswick, GA. In addition to these full stakeholder groups, we involved representatives from Georgia Department of Natural Resources (GADNR) and Georgia Department of Transportation (GDOT) as partial stakeholders that contributed expert knowledge of feasible management options and legal constraints as well as gave their values toward specific objectives. Authors of this study (BAC and CTM) acted as facilitators and technical experts who guided stakeholders through construction of the decision framework, TMN provided input as a representative of the GSTC (but was not involved in analysis), and JCM provided expert opinions of ecological science and management.

We elicited stakeholder input in three phases to develop the SDM framework: before, during, and after an all-stakeholder workshop. Although University of Georgia (UGA) and GSTC researchers had been collaboratively investigating road impacts and management since

2009, we formally initiated the decision making process in May of 2014. Through two in-person planning meetings, conference calls, and by email between UGA researchers and JIA staff (members of the Board of Directors and the GSTC), we led stakeholders in the development of a provisional problem statement, set of fundamental objectives, and list of actions to be taken off the table due to exorbitant legal, political, or financial costs. Next, we conducted a one-day workshop with the full team of representatives from each of the eight stakeholder groups in August 2014, facilitated by BAC and CTM. BAC began the workshop by presenting background information about the issue of road-related threats to terrapins, past research findings on the JIC, and the components and uses of SDM. The goal of the workshop was to iteratively discuss, define, and revise the scope of the problem, fundamental and means objectives, measurable attributes, and management alternatives by using products from the planning meetings as starting points to facilitate an efficient process. Following the workshop, we conducted a series of elicitation steps by email so that all stakeholders informed and approved the final set of objectives and management strategies to be evaluated in the decision framework.

Objectives

Representatives defined a set of five fundamental objectives important for management of the JIC: 1) maximize terrapin population persistence, 2) maximize human safety, 3) maximize patron satisfaction, 4) maximize transferable knowledge (learning), and 5) minimize net cost. These objectives spanned ecological, social, and economic dimensions of the conservation issue, and stakeholders recognized that some may compete or conflict with one another. For example, physical devices deployed on roadsides to increase terrapin survival and persistence may reduce road aesthetics and lead to decreased patron satisfaction. Facilitators guided representatives through the process of creating a hierarchy of objectives (i.e., an influence diagram: Figure 5.1) that separates fundamental objectives (those important in their own right) from means objectives (those that help achieve fundamental objectives) (Conroy & Peterson 2013). The team identified measurable attribute scales to evaluate achievement of each objective in the hierarchy (described in Appendix D). When natural scales (e.g., net cost measured in dollars) did not exist, the team created constructed scales based on agreed-upon "best" and "worst" states of an objective (Keeney & Gregory 2005). The terrapin population objective was implicit to existing efforts of multiple stakeholder groups (e.g., JIA, GSTC, GADNR, One Hundred Miles), and we specified the scale for this objective as the probability that the population would be decreasing (population growth rate [λ] < 0.98), stable ($0.98 < \lambda < 1.02$), or increasing ($\lambda > 1.02$) 50 years in the future. The human safety objective arose primarily from GDOT input, and all stakeholders agreed that any road management action has implications for user safety. We evaluated safety using a constructed scale with three levels: more, the same, or fewer vehicle accidents per year, relative to current rates on the JIC. Patron satisfaction, measured by percentage of road users satisfied, was also implicit in the mission of JIA, and stakeholders discussed its important implications to patron (visitor, resident, and employee) well-being as well as Jekyll Island's economy as a tourist destination. Transferable knowledge reflected the team's goals to learn from and apply actions tested on the JIC for other species and conservation contexts, and it was evaluated using a constructed scale reflecting the number of actions deployed and monitored on the JIC. Lastly, net cost of management decisions for the JIA and their partners was measured in dollars.

Stakeholders defined first- and second-order means objectives that influenced each fundamental objective (Figure 5.1). We briefly describe these relationships here, but see table headings in Appendix D for descriptions of attribute scales for means objectives. Terrapin persistence was influenced by the demographic processes of female survival and recruitment.

Survival was influenced by the level of per-capita road mortality for adult females, which is the only major threat to adult terrapins documented in the area (Crawford *et al.* 2014a); recruitment was influenced by the rate of success (i.e., not being depredated) for nests laid on JIC roadsides and the proportion of hatchlings born female. Safety, which considered both drivers and recreational cyclists on the JIC, depended on the number of people who walk across the road to intervene and remove terrapins (GSTC, pers. obs.), the likelihood of a driver maintaining their speed, trajectory, and overall control of their vehicle, and the presence of a bike path on the JIC (a potential management option described below). Level of vehicle control depended on the speed of travel (road efficiency), the proportion of drivers aware of risks on the road, and the number of terrapins (hazards) on the road. Number of interveners depended on the number of terrapins on the road that necessitate intervention. Satisfaction was influenced by speed of travel (road efficiency), road aesthetics, and conservation awareness. We note there was an inherent tradeoff where lower speed of travel was expected to increase safety but decrease patron satisfaction. Conservation awareness, distinct from awareness of risks, was described by stakeholders as the proportion of patrons who are aware and supportive of conservation efforts on Jekyll Island, which was expected to influence patron satisfaction due to Jekyll's status as a state park that attracts many visitors and residents who value the protection of local habitats and wildlife (Crawford, Poudyal & Maerz 2015). Transferable knowledge depended on the level of targeted research, use of devices applicable to other human-impacted populations of terrapins, and use of devices applicable to other species impacted by roads. Lastly, net cost depended on the direct costs to JIA and their partners, defrayed by the amount of external support generated.

Management alternatives

Stakeholders and experts co-developed 20 management alternatives (strategies) designed to address multiple road-associated threats. The diversity of stakeholder areas of expertise (e.g., road management, wildlife ecology, local infrastructure) facilitated the development of creative actions. The team first identified eight categories of management actions that collectively influenced the full set of objectives (Table 5.1). Within each category, team members brainstormed a list of potential management options that varied from “no management” or “current effort” to more intensive options that would require greater costs and/or be applied at larger scales. We briefly describe expected relationships between representative options in each category and objective outcome, but see Table D1 for the complete list of options and detailed descriptions. Traffic options influenced road efficiency (e.g., reducing the speed limit), where slower traffic was expected to increase drivers’ vehicle control. On-road signage was designed to increase driver awareness of risks while also affecting road aesthetics. Some signage options also influenced conservation awareness, such as the terrapin crossing signs with flashing warning beacons that have been programmed to flash during daily peaks of crossing activity through the nesting season since 2013 (see Chapter 4). Several types of barriers were included in our decision context to influence terrapins (number of terrapins on road, nest success), safety, and satisfaction (aesthetics). We used spatial peaks (hot spots) of terrapin-vehicle collisions on the JIC (Crawford *et al.* 2014b) to include barrier options at these discrete areas or broader scales. Only one barrier is currently deployed at one terrapin crossing hot spot on the JIC, which is a hybrid design that uses short fences to guide females into predator-proof nest boxes on elevated mounds (Buhlmann & Osborn 2011). Education options were designed to influence awareness of risks and conservation efforts, and included in-person outreach programs on Jekyll Island or broader dissemination of conservation messages using social media or local radio bulletins. Both predator and habitat management options influenced nest success and proportion of female hatchlings. Predator options included lethal removal or non-lethal, aversive conditioning (e.g., Conover 1990). Habitat management consisted of annual clearing of roadside hedges across varying proportions of the JIC, since nests laid in hedge habitats are more likely to be depredated and produce male hatchlings (Grosse *et al.* 2015). Research options influenced the degree of transferable knowledge gained. Lastly, financial options were designed to defray project costs.

We developed management strategies by selecting combinations of options across each category, similar to action portfolios (e.g., Blomquist *et al.* 2010). The team specified the following types of final strategies that focused on varying objectives:

1. Status quo (strategy 1): current management activities, including flashing warning signage, hybrid barrier at one hot spot, and current education and research activities from the GSTC.
2. Reduced effort (strategy 2): same actions as status quo but no further monitoring, research, or maintenance to reduce costs.
3. Tradeoff (strategies 3-9): most intensive/costly actions designed to meet one fundamental objective while not concerned with other objectives.
4. Compromise (strategies 10-12): moderate-intensity actions that were expected to be satisfactory over most objectives but not fully achieve any.
5. Hybrid (strategies 13-20): combinations of more intensive actions to meet one or two objectives and moderate actions for other objectives.

We present actions included in representative strategies in Table 5.2 and the full set of strategies in Appendix D (Table D4a).

Estimating consequences

We used a combination of empirical data measuring terrapin demographic rates, estimates from published literature, survey data measuring island patrons' support of management (Crawford, Poudyal & Maerz 2015), and expert opinion to model relationships between decisions and objective outcomes. Expert elicitation occurred following the SDM workshop, and we averaged and shared responses with the expert group and stakeholder team for final approval. Estimates for the effects of actions on objectives related to terrapin persistence came from eleven professional scientists with terrapin research experience from across the species range. For model components related to safety, satisfaction, and cost objectives, we elicited expert opinion from subsets of six representatives from stakeholder groups (e.g., GDOT, JIA, GSTC, residents) familiar with local management, education, and patrons' perceptions of the JIC and management. These experts provided two types of input used to parameterize the decision network: 1) conditional probability tables representing the likelihood a certain objective outcome given an action or outcome of a lower-level objective, and 2) direct estimates of consequences of single actions or combinations of actions included in the list of management strategies. To limit the response burden, we often asked for estimates for a range of actions or objective states but not every action, combination, or objective state. We then used the range of estimates to interpolate conditional probabilities and action consequences for cases not directly estimated by experts. We parameterized model components in the network, while incorporating uncertainty, by using mean conditional probabilities or using the means and standard deviations from respondents' direct estimates to calculate appropriate probability distributions. See Appendix D for conditional probability tables and estimated consequences of actions and strategies.

We developed a predictive model to estimate outcomes for the terrapin persistence objective under each management strategy using population viability analysis (Akçakaya & Sjögren-Gulve 2000). Briefly, we modeled the effects of the following types of management actions on means objectives linked to persistence: 1) on-road signage, speed reductions, and awareness campaigns influenced survival of crossing females, 2) roadside barriers influenced the number of nesting females crossing the road, 3) predator removal influenced nest success, and 4) roadside vegetation management influenced nest success and proportion of hatchlings born female. We estimated the effects of flashing warning signage and the hybrid barrier using empirical data (see Chapter 4), estimated the effects of vegetation clearing using empirical data (see Chapter 2), and estimated all other effects using published literature and terrapin expert opinion. We then repeatedly projected the population for 50 years under each management strategy, using methods that incorporate parametric uncertainty and annual stochasticity (e.g., Moore *et al.* 2012), and calculated the likelihood the population was declining, stable, or increasing (see Chapter 4 for detailed description of modeling procedures).

We used expert opinion to model the effects of the following types of actions on means objectives influencing safety and satisfaction: signage and education alternatives on driver awareness of risks; signage, education, predator management, and financial alternatives on patron awareness and support for conservation efforts; and signage (including interpretive signs), barriers, and vegetation management alternatives on road aesthetics. To inform estimates of conservation awareness and support, we used survey data obtained from > 1300 island patrons (see Chapter 3) that showed the percentage of patrons that deemed the following management actions as acceptable: flashing warning signage, short fences, nest boxes, vegetation clearing, and lethal predator removal. We used this as a proxy measure of conservation support and averaged these values with expert estimates to compute final percentages of patrons supportive given each action. Mean estimates from survey data and expert opinion tended to agree; differences ranged from 1% (lethal predator removal) to 14% (flashing warning signage). We had experts estimate outcomes of each objective under current, baseline conditions, which allowed the estimation of the effect size of each action on the objective. For strategies that included multiple actions that influenced a single objective, we assumed effects were additive when computing the cumulative objective outcome from each strategy.

We used simple, deterministic models for road efficiency and objectives influencing transferable knowledge and net cost. We assigned the level of road efficiency under each strategy based on the traffic actions included in each strategy (high = current speed limit, moderate = increased enforcement, low = reduced speed limit). For objectives influencing transferable knowledge, we assigned each strategy a categorical score based on the number of actions it included that satisfied each objective (1 = current actions, 2 = one additional action, 3 = more than one additional action). For example, a strategy that included speed displays and predator removal would receive a score of 3 for the "actions transferable to other species" objective since these actions have been used for other species (e.g., Sullivan *et al.* 2004; Beasley *et al.* 2013). The scale for targeted research also included a 0 value, indicating reduced research effort. BAC assigned scores, which were agreed upon by stakeholders, and we summed scores for each objective to compute the categorical score for transferable knowledge (from 2 [reduced effort] to 9 [maximum research and actions applied]). Lastly, we used online resources and personal communications with stakeholders (JIA, GSTC) to estimate the direct costs and external funds for each action and summed costs of all actions within each strategy (Table D3e).

Evaluating optimal decisions

We modeled relationships between management strategies and objective outcomes using a Bayesian belief network (Figure 5.2) implemented in Netica (www.norsys.com), which provides a graphical representation of relationships (similar to an influence diagram) and allows for Bayesian updating (Conroy & Peterson 2013). In decision analysis, the expected value of a decision is the product of three components: the stakeholder's relative importance placed on each objective (i.e., objective weights), the value placed on outcomes within each objective (i.e., utilities), and the probability of those outcomes occurring (Clemen & Reilly 2013).

Given the multiple objectives within the framework that may compete or conflict, we used the swing weighting technique (Goodwin & Wright 2007) for each stakeholder group to assign relative importance to each of the five fundamental objectives. Although the team's objectives focused on types of outcomes that are not easily commensurable (e.g., terrapin persistence, patron satisfaction, cost), swing weighting provides a method for valuating objectives on a common scale that recognizes some objectives and outcomes are more important and desirable than others. Following the workshop, we distributed swing weighting exercises for representatives to complete. We first asked groups to consider a scenario where all fundamental objectives were at their least-preferred level (i.e., declining population, more vehicle accidents, patron dissatisfaction, no gain of knowledge, and high project costs). Representatives then ranked objectives by which one they would most or least like to improve. Representatives assigned a rating from 0 to 100 for each objective, where the top-ranked objective received a value of 100. These ratings allowed stakeholders to distribute value across the set of objectives with finer precision. We normalized each respondent's ratings and averaged these scores across stakeholder groups to calculate mean weights, W_j, for each objective j, and weights summed to 1.

For groups with more than one representative participating, we averaged weights within a group before averaging across all groups, so that each group's responses had equal weight. We also retained objective weights for each stakeholder group i, $W_{i,j}$, to later examine if optimal decisions varied across stakeholders (see below).

Within each objective, representatives rated the utility (desirability) of each outcome level x_k (e.g., the terrapin population is declining, stable, or increasing) using a 0 to 100 scale. We then normalized raw scores to a 0-1 scale using proportional scoring (see Conroy & Peterson 2013), where utilities are calculated as

$$U(x_k) = \frac{x_k - worst(x_k)}{best(x_k) - worst(x_k)}$$

and best(x_k) and worst(x_k) are the most and least desirable levels within that objective. This process allowed stakeholders to specify nonlinear relationships between achieving different levels of an objective. For example, stakeholders assigned a mean utility value of 0.64 for achieving a stable terrapin population, and not 0.50 as would have been assigned using a linear relationship between the utilities for a declining (0) and increasing (1) population. We averaged ratings across stakeholders using the same procedures described above. We summarized and distributed objective weights and utilities to the team so that stakeholders could adjust and approve their responses before we evaluated decisions.

In addition to fundamental objectives, stakeholders performed the same swing weighting and outcome utility procedures for the three means objectives influencing patron satisfaction: road efficiency, road aesthetics, and conservation awareness. We expected that relative weights may vary substantially between stakeholder groups. In order to more accurately represent the values of large public stakeholder groups (island visitors, residents, and employees), we averaged the objective weights assigned by these groups' representatives with patron values elicited from survey-based research (Crawford, Poudyal & Maerz 2015). Specifically, we used the degree to which surveyed patrons in each group agreed with the statements, “We should maintain the current speed limit regardless of the impact on terrapins”, “We should maintain views of the marshland from the Causeway regardless of the impact on terrapins”, and “Management should focus on doing what is best for terrapins over what is best for island patrons”, as relative weights given to road efficiency, aesthetics, and conservation awareness and support, respectively.

We computed the expected value for each management strategy l (D_l) in Netica and selected optimal decisions using a weighted utility function:

$$E(D_l) = \sum_j \sum_k W_j * U\left(x_{j,k}\right) * p\left(x_{j,k,l}\right).$$

The function summed over all fundamental objectives (j) and outcome levels (k) within each objective the product of the objective’s weight, the outcome’s utility, and the probability (p) of that outcome occurring, given each strategy. We used this same function to calculate the level of patron satisfaction, which incorporated the relative weights and utilities of objective outcomes for its three means objectives.

We evaluated the robustness of optimal decisions to existing uncertainties in the decision framework using sensitivity analysis (Clemen & Reilly 2013). In order to identify model components that had the greatest influence on the expected value of the decision, as well as which decision ranked as best, we performed a one-way sensitivity analysis by systematically varying each model component (means objectives) influencing safety and satisfaction to their minimum and maximum levels, in turn. We evaluated the sensitivity of terrapin persistence in Chapter 4. We did not evaluate sensitivity of decisions to road efficiency or objectives influencing transferable knowledge gained and net cost since states of these objectives were deterministic outcomes of each management strategy. Lastly, we examined the sensitivity of optimal decisions to variation in stakeholder values. We updated the decision model by replacing average weights assigned to the five fundamental objectives and three means objectives influencing patron satisfaction with the weights assigned by each of the six full stakeholder groups, in turn, and we recorded the best- and worst-performing management strategies.

RESULTS

On average, stakeholders assigned the highest weight to maximizing terrapin persistence, followed by maximizing human safety, and lowest weight to maximizing transferable knowledge (Table 5.3). For objectives influencing patron satisfaction, stakeholders assigned roughly equal weight to conservation awareness and road aesthetics and less weight to road efficiency. The weight and relative rank of objectives varied by stakeholder (Table 5.3).

When considering the full suite of objectives, the best-performing management strategy, indicated by highest expected value, was strategy 20 Hybrid, followed by 17 Hybrid and 3 Tradeoff (Table 5.4). Hybrid strategies, which focused heavily on one objective with complementary actions targeting other objectives, tended to rank highly. The top hybrid strategy was predicted to result in a 59.5% chance the terrapin population would be stable or increasing, a 97.1% chance safety would remain at current or improved levels, and 74.5% patron satisfaction. As expected, Tradeoff strategies performed variably well, receiving high or low rankings, and Compromise strategies ranked toward the middle of the pool. The worst strategy was 1 Status quo, representing current management conditions. The best-performing strategy differed when considering each objective in isolation. Results from the terrapin population model showed that the probability the population will be stable or increasing ($\lambda > 0.98$) within the next 50 years ranged widely from 0.05, given current management conditions and less monitoring effort (2

Reduced effort), to 0.85, given a strategy that included the most intensive and costly options for roadside barriers, predator removal, and vegetation clearing (3 Tradeoff). All strategies yielded a high (> 0.95) probability that vehicle accidents would remain at or below current levels. Expected patron satisfaction ranged from 48% (strategy 3 Tradeoff) to 85% (15 Hybrid). The set of strategies yielded all possible outcomes on the scale for transferable knowledge gained. Finally, net cost for strategies ranged from $110,000 (9 Tradeoff) to $795,000 (3 Tradeoff). We found certain strategies were dominated, meaning another strategy performed better than the dominated strategy on one objective and equal or better on all other objectives (Conroy & Peterson 2013). Strategy 13 Hybrid was dominated by 17 Hybrid and 19 Hybrid; 11 Compromise was dominated by 10 Compromise.

The same management strategy (20 Hybrid) ranked first among the list of strategies with the highest expected value regardless of changes made to objective levels during the one-way sensitivity analysis. Expected value was most sensitive to the road aesthetic state and patron conservation awareness and least sensitive to driver risk awareness (Figure 5.3). Strategy 20 Hybrid also ranked first with the highest expected value when we parameterized the model using mean or stakeholder-specific objective weights (Table 5.3). The status quo strategy always performed worst regardless of objective weights used.

DISCUSSION

Developing an SDM framework for multiple stakeholders to navigate a complex road management problem revealed findings essential for decision making for the JIC: (i) stakeholders valued maximizing terrapin persistence, followed closely by human safety, above other objectives within the decision context, (ii) maintaining the status quo was the least preferred strategy, giving impetus for additional management, and (iii) potential strategies varied in their expected outcomes across objectives, but preferred strategies focused on intensive terrapin management while also including complementary actions aimed at meeting socioeconomic objectives.

The systematic approach of SDM is particularly suited for challenging conservation issues like road management where multiple, socioecological objectives must be valued and compared in order to identify optimal decisions (e.g., Converse *et al.* 2013). However, this process requires careful consideration of the nature of these objectives and how they are compared. Different objectives may be incommensurable (e.g., maximizing wildlife persistence or human well-being) to the point that pooling them into a single metric fails to capture what different stakeholders truly value (Hirsch *et al.* 2011; Vercoe *et al.* 2014). Still, developing a metric (i.e., a weighted average) for the degree to which strategies meet a full suite of objectives as well as single objectives is useful for assessing tradeoffs within a decision context and producing recommendations. To compute average weights, stakeholders recognize that certain outcomes for each objective are more preferred than others and place value on each objective in a concrete way specific to the decision context, local attitudes, and the range of possible outcomes for each objective (Goodwin & Wright 2007). For example, stakeholders rated terrapin persistence as more important, on average, than human safety, but this does not reflect that stakeholders valued saving wildlife more than keeping humans safe. Instead, the results show that it was more important for management to shift the terrapin population trajectory from declining to increasing than to decrease the number of accidents per year on the JIC. The importance assigned to terrapin persistence complements previous findings that Jekyll Island patrons held biocentric values and supported terrapin management (Crawford, Poudyal & Maerz 2015). It is also reasonable that human safety did not outrank terrapin persistence since stakeholders likely viewed any terrapin-vehicle accident as non-lethal for motorists. Although terrapins, like any object on the road, present a potential safety hazard if drivers must swerve to avoid them, human safety will likely be more important in other road management contexts involving large ungulates that pose a larger risk of fatality (e.g., Putman 1997). Ultimately, the procedures for weighting multiple objectives ensured that stakeholder values were represented accurately and transparently in the decision model and provided a means to identify which decisions were optimal on average as well as for specific objectives or stakeholder groups (Table 5.4).

We recognize that some fundamental objectives within our framework may be partially redundant (i.e., correlated: Clemen & Reilly 2013), which may inflate the expected values of certain strategies. Terrapin persistence and human safety objectives were both strongly influenced by the means objective of minimizing the number of terrapins on the road. The terrapin population viability model was most sensitive to proportion of females crossing (Chapter 4), and the decision model was more sensitive to number of terrapins on the road than all other means objectives influencing human safety. Thus, strategies that included barriers preventing terrapins from accessing the road had positive effects on persistence and human safety. However, terrapin persistence and human safety represented distinct goals that were influenced by non-overlapping means objectives, and our framework included actions that could independently affect each objective. Partial redundancy also existed between terrapin persistence and patron satisfaction objectives. As noted previously, Jekyll Island attracts patrons that tend to value the protection of wildlife, so efforts to maintain terrapin persistence may also increase patron satisfaction. This relationship was included in our framework as the means objective for conservation awareness and support. However, the decision maker (JIA) viewed terrapin persistence and patron satisfaction objectives as distinct and fundamental to their mission. Through early stages of the all-stakeholder workshop, representatives acknowledged that maximizing terrapin persistence was important for other reasons separate from satisfaction (e.g., maintaining natural integrity of Jekyll Island), and satisfaction was heavily influenced by other factors (speed of travel and road aesthetics) not associated with efforts to conserve terrapins. Ultimately, including terrapin persistence, safety, and satisfaction as separate objectives allowed for a greater understanding of the JIC issue that is likely transferable to other road management contexts.

When considering all objectives, the decision model indicated that maintaining current management actions was the worst strategy, and implementing additional measures was necessary for JIC management. The two best-performing alternatives were Hybrid strategies containing similar actions designed to address the terrapin population objective as well as partially satisfy other objectives. Actions included a bike path and additional short fence barriers, interpretive or other on-road signage to increase risk and conservation awareness, intensive predator removal, and moderate vegetation management to increase road aesthetics. We observed tradeoffs between objectives within our decision context as neither strategy performed best for any one objective; instead, they performed well for most objectives (Table 5.4). In contrast, the third-ranked strategy (3 Tradeoff) resulted in the highest outcome for terrapin persistence (the most important objective by weight) but resulted in the lowest level of patron satisfaction and highest cost of any alternative. Other Tradeoff strategies that focused on a single objective or management category while ignoring others performed poorly. These results support previous conservation studies showing that complementary actions are often the best means for addressing multiple threats and objectives in complex systems without requiring any threat be completely eliminated or objective be fully satisfied (Gregory & Long 2009; Converse *et al.* 2013; Crawford *et al.* 2014a).

The SDM process provided a venue for stakeholder groups to better understand each other's perspectives, contribute heterogeneous sources of knowledge, and build trust among parties who either collaborate infrequently or may work at cross purposes (Pretty & Smith 2004). Maintaining positive public relations is a fundamental goal for JIA, who must consider often-diverging preferences among stakeholders and the socioeconomic outcomes for Jekyll Island. Despite stakeholder groups varying in which objectives were deemed most important for JIC management, all stakeholders, including public groups (e.g., visitors and residents), constructively engaged throughout the decision making process. Furthermore, all stakeholders were given an equal "voice" when modeling the decision. Not only did we weight stakeholders' preferences equally when computing average objective weights, but the overall best-performing strategy was also most preferred when we used each stakeholder's values, in turn, in the model. Thus, the optimal strategy represented a truly shared preference among the team. However, other contexts may require decision makers to give more weight to the values of certain stakeholders based on group size or political influence. Ultimately, using SDM to involve stakeholders and transparently incorporating their preferences toward management actions should circumvent policy delays and wasting of resources (Granek *et al.* 2008; Gregory & Long 2009).

We structured the current framework as a one-time decision since many actions considered were permanent (e.g., roadside barriers, on-road signage). However, this and other applications of SDM for road management could incorporate monitoring of seasonal actions (e.g., predator removal, vegetation clearing, awareness programs) to reduce uncertainties and improve decisions using an adaptive management framework (Conroy & Peterson 2013).

Substantial uncertainty of terrapin population dynamics remained in the decision model and should be resolved with future monitoring. Despite being well-studied, estimating population parameters of diamondback terrapins in this and other studies yielded a high degree of uncertainty (see Chapter 4) due to their cryptic life stages, low rates of detection, long generation times, and dynamic movements through complex habitats (Butler, Heinrich & Seigel 2006). Uncertainty was compounded in population viability models as we used expert opinions for the effects of many management actions. Establishing population monitoring programs should help replace expert opinions with empirical data and lead to more accurate decision models. We also acknowledge that management decisions for other terrapin populations will need to consider additional threats operating currently (e.g., crab pot mortality, habitat development) or those that may emerge in the future (e.g., sea level rise impacts on marsh habitat) not addressed in this study. The SDM process is sufficiently flexible to accommodate case-specific threats and changing information, and our approach could be adapted to inform decisions for declining terrapin populations range-wide.

Future applications of SDM should incorporate human dimensions data in place of expert opinion when estimating social outcomes from proposed management actions. Our study is one of the first to integrate human dimensions research alongside population models in an SDM framework, and it demonstrates how this novel practice can yield more accurate and complete predictions of social and ecological outcomes. Predicting the effects of strategies on objectives such as public satisfaction can be coarsely estimated by a small group of representatives in a workshop, but obtaining accurate estimates will require survey-based data of a broader sample of large, key stakeholder groups (e.g., Vaske & Donnelly 2007). Elicitation of groups' values and attitudes can be conducted easily using online or in-person surveys (Vaske 2008), and this process should be informed by consultation with experts in human dimensions and qualitative research from the social sciences. Surveys can be used to monitor expected or unexpected shifts in attitudes following implementation of management. For the JIC, conducting further survey-based research of patrons' attitudes can estimate the effectiveness of signage and awareness programs and be used to modify program features to increase pro-conservation attitudes and acceptance of management strategies. Additionally, the two most preferred strategies included lethal raccoon removal, which was the most controversial potential action with surveyed patrons (Crawford, Poudyal & Maerz 2015). Strategies with predator management resulted in higher population persistence, which was the most important objective for stakeholders, but further surveying and dialogue with patron groups will likely be necessary to measure and reduce any negative impact this action has on public satisfaction. As JIA and local partners consider the recommendations of this study and implement actions on the JIC, continued monitoring of patron attitudes will allow empirical data on management effects to inform these estimates. Given that most conservation issues require managers to consider social objectives and the preferences of diverse groups, including the general public, integrating human dimensions research should benefit applications of SDM by enhancing the understanding of an issue's social dimensions and ensuring optimal decisions reflect this information.

ACKNOWLEDGEMENTS

This research was supported in part by the Daniel B. Warnell School of Forestry and Natural Resources, University of Georgia and AGL Resources Foundation through the Jekyll Island Foundation, which raises funds for projects related to the conservation, preservation and education of Jekyll Island. We thank all stakeholder representatives for their committed support throughout this research, including members of the Jekyll Island State Park Authority, Georgia

Sea Turtle Center, Jekyll Island Citizens Association, Golden Isles Convention and Visitors Bureau, Georgia Department of Natural Resources, Georgia Department of Transportation, and One Hundred Miles. We especially thank B. Carswell and M. Kaylor for coordination and feedback. All methods were conducted with the approval of the University of Georgia Institutional Review Board (Project #: 2012-10815-1).

REFERENCES

Akçakaya, H.R. & Sjögren-Gulve, P. (2000) Population viability analyses in conservation planning: an overview. *Ecological Bulletins***,** 9-21.

Andrews, K.M., Nanjappa, P. & Riley, S.P.D. (2015) *Roads and Ecological Infrastructure: Concepts and Applications for Small Animals*. Johns Hopkins University Press, Baltimore, MD.

Aresco, M.J. (2005) Mitigation measures to reduce highway mortality of turtles and other herpetofauna at a north Florida lake. *Journal of Wildlife Management,* **69,** 549-560.

Baird, R.C. (2009) Coastal urbanization: the challenge of management lag. *Management of Environmental Quality: An International Journal,* **20,** 371-382.

Beasley, J.C., Olson, Z.H., Beatty, W.S., Dharmarajan, G. & Rhodes Jr, O.E. (2013) Effects of culling on mesopredator population dynamics. *PloS one,* **8,** e58982.

Beaudry, F., Demaynadier, P.G. & Hunter, M.L. (2010) Identifying hot moments in road-mortality risk for freshwater turtles. *Journal of Wildlife Management,* **74,** 152-159.

Blomquist, S.M., Johnson, T.D., Smith, D.R., Call, G.P., Miller, B.N., Thurman, W.M., McFadden, J.E., Parkin, M.J. & Boomer, G.S. (2010) Structured decision-making and rapid prototyping to plan a management response to an invasive species. *Journal of Fish and Wildlife Management,* **1,** 19-32.

Buhlmann, K.A. & Osborn, C.P. (2011) Use of an artificial nesting mound by wood turtles (*Glyptemys insculpta*): a tool for turtle conservation. *Northeastern Naturalist,* **18,** 315-334.

Bureau, U.S.C. (2010) American Fact Finder. Available from http://factfinder2.census.gov/faces/nav/jsf/pages/community_facts.xhtml (accessed on 5 May 2012).

Butler, J., Heinrich, G. & Seigel, R. (2006) Third workshop on the ecology, status, and conservation of diamondback terrapins (*Malaclemys terrapin*): results and recommendations. *Chelonian Conservation and Biology,* **5,** 331-334.

Clemen, R. & Reilly, T. (2013) *Making Hard Decisions with Decision Tools,* Third edn. Cengage Learning, Boston, MA.

Conover, M.R. (1990) Reducing mammalian predation on eggs by using a conditioned taste aversion to deceive predators. *The Journal of Wildlife Management***,** 360-365.

Conroy, M.J. & Peterson, J.T. (2013) *Decision making in natural resource management: a structured, adaptive approach*. Wiley,

Converse, S.J., Moore, C.T., Folk, M.J. & Runge, M.C. (2013) A matter of tradeoffs: reintroduction as a multiple objective decision. *The Journal of Wildlife Management,* **77,** 1145-1156.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A. & Norton, T.M. (2014a) Estimating the consequences of multiple threats and management strategies for semi-aquatic turtles. *Journal of Applied Ecology,* **51,** 359-366.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A., Norton, T.M. & Albeke, S.E. (2014b) Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology,* **51,** 367-375.

Crawford, B.A., Poudyal, N.C. & Maerz, J.C. (2015) When Drivers and Terrapins Collide: Assessing Stakeholder Attitudes Toward Wildlife Management on the Jekyll Island Causeway. *Human Dimensions of Wildlife,* **20,** 1-14.

Cureton, J.C. & Deaton, R. (2012) Hot moments and hot spots: identifying factors explaining temporal and spatial variation in turtle road mortality. *Journal of Wildlife Management,* **76,** 1047-1052.

Fahrig, L. & Rytwinski, T. (2009) Effects of roads on animal abundance: an empirical review and synthesis. *Ecology and Society,* **14,** 21.

Feinberg, J.A. & Burke, R.L. (2003) Nesting ecology and predation of diamondback terrapins, *Malaclemys terrapin*, at Gateway National Recreation Area, New York. *Journal of Herpetology,* **37,** 517-526.

Game, E.T., Meijaard, E., Sheil, D. & McDonald-Madden, E. (2014) Conservation in a Wicked Complex World; Challenges and Solutions. *Conservation Letters,* **7,** 271-277.

Georgia Department of Transportation (2014) Georgia's State Traffic and Report Statistics (STARS). Available from http://www.dot.state.ga.us/statistics/stars/Pages/GlynnTraffic.aspx (accessed on 19 Oct 2011).

Glista, D.J., DeVault, T.L. & DeWoody, J.A. (2009) A review of mitigation measures for reducing wildlife mortality on roadways. *Landscape and Urban Planning,* **91,** 1-7.

Goodwin, P. & Wright, G. (2007) *Decision analysis for management judgment*. John Wiley & Sons, Chichester, United Kingdom.

Granek, E., Madin, E., Brown, M., Figueira, W., Cameron, D., Hogan, Z., Kristianson, G., De Villiers, P., Williams, J. & Post, J. (2008) Engaging recreational fishers in management and conservation: global case studies. *Conservation Biology,* **22,** 1125-1134.

Gregory, R., Failing, L., Harstone, M., Long, G., McDaniels, T. & Ohlson, D. (2012) *Structured decision making: a practical guide to environmental management choices*. John Wiley & Sons,

Gregory, R. & Long, G. (2009) Using structured decision making to help implement a precautionary approach to endangered species management. *Risk Analysis,* **29,** 518-532.

Grosse, A.M., Crawford, B.A., Maerz, J.C., Buhlmann, K.A., Norton, T.M., Kaylor, M. & Tuberville, T.D. (2015) Effects of vegetation structure and artificial nesting habitats on hatchling sex determination and nest survival of diamondback terrapins. *Journal of Fish and Wildlife Management,* **6,** 19-28.

Hirsch, P.D., Adams, W.M., Brosius, J.P., Zia, A., Bariola, N. & Dammert, J.L. (2011) Acknowledging conservation trade-offs and embracing complexity. *Conservation Biology,* **25,** 259-264.

Keeney, R. L. & Gregory, R. S. (2005) Selecting attributes to measure the achievement of objectives. *Operations research*, **53**, 1-11.

Liu, S., Walshe, T., Long, G. & Cook, D. (2012) Evaluation of Potential Responses to Invasive Non-Native Species with Structured Decision Making. *Conservation Biology,* **26,** 539-546.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L. & Pulgar-Vidal, M. (2011) Hard choices:

Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation,* **144,** 966-972.

Moore, C.T., Converse, S.J., Folk, M.J., Runge, M.C. & Nesbitt, S.A. (2012) Evaluating release alternatives for a long-lived bird species under uncertainty about long-term demographic rates. *Journal of Ornithology,* **152,** 339-353.

Nichols, J.D. & Williams, B.K. (2006) Monitoring for conservation. *Trends in Ecology & Evolution,* **21,** 668-673.

Pretty, J. & Smith, D. (2004) Social capital in biodiversity conservation and management. *Conservation Biology,* **18,** 631-638.

Putman, R.J. (1997) Deer and road traffic accidents: Options for management. *Journal of Environmental Management,* **51,** 43-57.

Rhodes, J.R., Ng, C.F., de Villiers, D.L., Preece, H.J., McAlpine, C.A. & Possingham, H.P. (2011) Using integrated population modelling to quantify the implications of multiple threatening processes for a rapidly declining population. *Biological Conservation,* **144,** 1081-1088.

Stout, R.J., Stedman, R.C., Decker, D.J. & Knuth, B.A. (1993) Perceptions of risk from deer-related vehicle accidents: implications for public preferences for deer herd size. *Wildlife Society Bulletin,* **21,** 237-249.

Sullivan, T.L., Williams, A.F., Messmer, T.A., Hellinga, L.A. & Kyrychenko, S.Y. (2004) Effectiveness of temporary warning signs in reducing deer-vehicle collisions during mule deer migrations. *Wildlife Society Bulletin,* **32,** 907-915.

Szerlag-Egger, S. & McRobert, S.P. (2007) Northern diamondback terrapin occurrence, movement, and nesting activity along a salt marsh access road. *Chelonian Conservation and Biology*, **6,** 295-301.

Szerlag, S. & McRobert, S.P. (2006) Road occurrence and mortality of the northern diamondback terrapin, *Malaclemys terrapin terrapin,* during the nesting season. *Applied Herpetology,* **3,** 27-37.

Trombulak, S.C. & Frissell, C.A. (2000) Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology,* **14,** 18-30.

Vaske, J.J. (2008) *Survey research and analysis: Applications in parks, recreation and human dimensions*. Venture Publishing, Inc., State College, PA.

Vaske, J.J. & Donnelly, M.P. (2007) *Public knowledge and perceptions of the desert tortoise*. Colorado State University, Fort Collins, CO.

Vercoe, R.A., Welch-Devine, M., Hardy, D., Demoss, J.A., Bonney, S.N., Allen, K., Brosius, P., Charles, D., Crawford, B., Heisel, S., Heynen, N., de Jesús-Crespo, R.G., Nibbelink, N., Parker, L., Pringle, C., Shaw, A. & Van Sant, L. (2014) Acknowledging Trade-offs and Understanding Complexity: Exurbanization Issues in Macon County, North Carolina. *Ecology and Society,* **19**, 23.

Table 5.1. Representative management actions across eight categories and their relationships with objectives included in the structured decision making framework for management of the Jekyll Island Causeway, Jekyll Island, GA. Note: see Appendix D for full list of actions and relationships to objectives.

Management categories	**Representative Actions**	**Objective(s) impacted**
Traffic	Inc. speed enforcement Reduced speed (45mph)	Road efficiency
Signage	Flashing terrapin crossing warning signs Digital speed displays	Risk awareness, conservation awareness, road aesthetics
Barriers	Hybrid barriers (short fences and predator-proof nest boxes) Bike path with curb-style barrier	Number of terrapins on road, nest success, conservation awareness, road aesthetics, safety (bike path only)
Habitat	Roadside hedge clearing (various degrees)	Nest success, proportion of female hatchlings, conservation awareness, road aesthetics
Predators	Non-lethal aversive conditioning Lethal removal (locally or JIC-wide)	Nest success, conservation awareness
Education	Social media messaging Interpretive signage on JIC	Risk awareness, conservation awareness, road aesthetic (interpretive signage only)
Research	Monitoring of effects of fencing or predator options	Targeted research
Financial	Fundraisers for local donors Conservation fees for road users	Conservation awareness, external support

Table 5.2. Representative management strategies and included actions.

		Management category							
Strategy # (type)	**Description**	**Traffic**	**Signage**	**Barriers**	**Habitat**	**Predators**	**Education**	**Research**	**Financial**
1 Status quo	Current mgmt. actions	55mph	Current (flashing warning signs)	Current (hybrid barrier at 1 hot spot)	Current roadside hedge clearing	None	Current	Current	None
4 Tradeoff	Intensive barriers for terrapin persistence	55mph	Current	Bike path (north & south side)	Current	None	Current	Barriers	None
11 Compromise	Low cost, localized terrapin mgmt.	Inc. enforcement	Current, speed display	Short fences (3 hot spots)	Moderate roadside hedge clearing	Localized aversive conditioning	Social media, JI programs	Predators	Grants
17 Hybrid	Moderate terrapin mgmt., risk and conservation awareness actions	55mph	Current, speed display, scoreboard	Bike path (north side), terrapin gardens (3 hot spots, south side)	Moderate roadside hedge clearing	JIC-wide lethal removal	Interpretive signage	Barriers, predators	Fees for JIC users, grants

Table 5.3. Mean and stakeholder-specific weights[a] assigned to each of 5 fundamental objectives and 3 means objectives influencing patron satisfaction, and the best- and worst-ranked management strategies given objective weights, relevant to management of the Jekyll Island Causeway, Jekyll Island, GA.

	Fundamental objectives					**Satisfaction objectives**			**Rank[b]**			
Stakeholder	**Terrapins**	**Safety**	**Satisfaction**	**Knowledge**	**Cost**	**Road efficiency**	**Road aesthetics**	**Cons. awareness**	**1**	**2**	**3**	**Worst**
Overall mean	0.25	0.24	0.18	0.16	0.18	0.29	0.35	0.36	20 H	17 H	3 T	1 SQ
1	0.24	0.29	0.06	0.15	0.26	0.30	0.45	0.25	20 H	3 T	17 H	1 SQ
2	0.26	0.23	0.14	0.18	0.19	0.28	0.26	0.46	20 H	3 T	17 H	1 SQ
3	0.19	0.22	0.24	0.21	0.14	0.24	0.37	0.39	20 H	17 H	19 H	1 SQ
4	0.31	0.15	0.20	0.16	0.17	0.34	0.32	0.34	20 H	3 T	17 H	1 SQ
5	0.22	0.27	0.24	0.11	0.16	0.31	0.34	0.35	20 H	17 H	19 H	1 SQ
6	0.27	0.25	0.18	0.14	0.16	0.23	0.30	0.47	20 H	3 T	17 H	1 SQ

[a] Respondents originally rated objectives between 0 (least important) and 100 (most important), and scores were normalized for each respondent and averaged across stakeholder groups.

[b] Letters indicate strategy types: H = Hybrid; T = Tradeoff; SQ = Status quo.

Table 5.4. Rank, expected value, and predicted outcomes for the five fundamental objectives for potential management strategies for the Jekyll Island Causeway, Jekyll Island, GA. For terrapin population and safety objectives, we present outcomes as the probability the most desirable state occurs. Stakeholders desired to minimize cost and maximize all other objectives, with lighter shades indicating more desired outcomes.

			Fundamental objectives				
Strategy	**Rank**	**Expected value**	**Terrapins (prob of λ > 0.98)**	**Safety (prob of reduced accidents)**	**Satisfaction (%)**	**Transferable knowledge (2-9 scale)**	**Net Cost (hundred thousand $)**
20 (Hybrid)	1	0.698	0.595	0.722	74.5	8	5.140
17 (Hybrid)	2	0.661	0.402	0.761	74.6	8	4.841
3 (Tradeoff)	3	0.659	0.850	0.925	47.9	7	7.954
19 (Hybrid)	4	0.649	0.392	0.692	75.5	9	4.857
8 (Tradeoff)	5	0.643	0.389	0.212	73.7	9	3.499
5 (Tradeoff)	6	0.633	0.406	0.230	64.9	8	3.199
16 (Hybrid)	7	0.633	0.417	0.684	61.3	9	4.711
10 (Compromise)	8	0.583	0.225	0.226	54.7	8	2.169
14 (Hybrid)	9	0.583	0.205	0.709	74.4	7	4.506
12 (Compromise)	10	0.574	0.064	0.410	59.2	5	1.985
15 (Hybrid)	11	0.572	0.104	0.656	84.6	7	4.436
18 (Hybrid)	12	0.568	0.116	0.664	80.6	7	4.454
13 (Hybrid)	13	0.554	0.210	0.620	68.2	7	4.863
11 (Compromise)	14	0.550	0.119	0.215	51.9	8	2.511
4 (Tradeoff)	15	0.533	0.410	0.896	67.3	5	7.043
7 (Tradeoff)	16	0.524	0.064	0.179	83.4	4	2.243
6 (Tradeoff)	17	0.521	0.311	0.799	49.4	5	5.312
2 (Reduced effort)	18	0.462	0.045	0.150	69.7	2	1.546
9 (Tradeoff)	19	0.458	0.047	0.150	67.2	3	1.099
1 (Status quo)	20	0.435	0.048	0.143	69.7	3	1.835

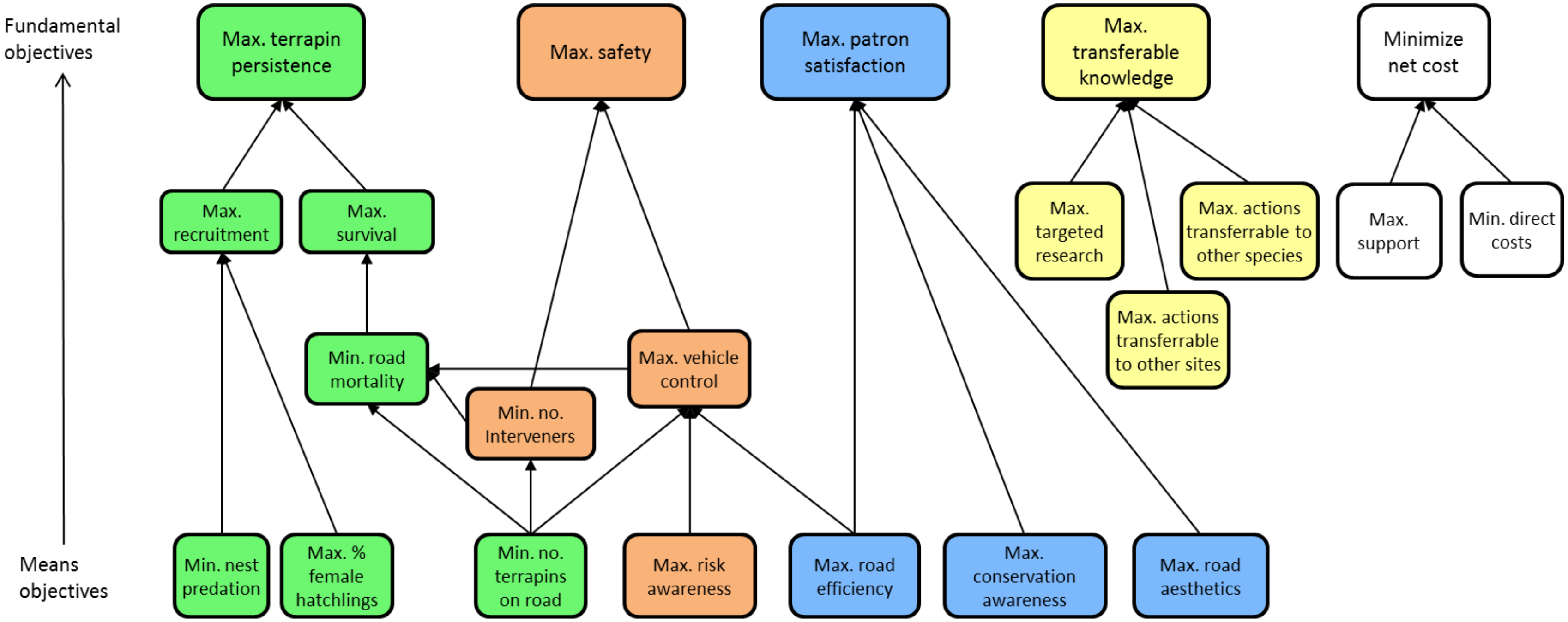


Figure 5.1. Objective hierarchy network showing relationships (arrows) between fundamental (top level) and means (lower levels) objectives for conservation management of diamondback terrapins on the Jekyll Island Causeway, Jekyll Island, GA.

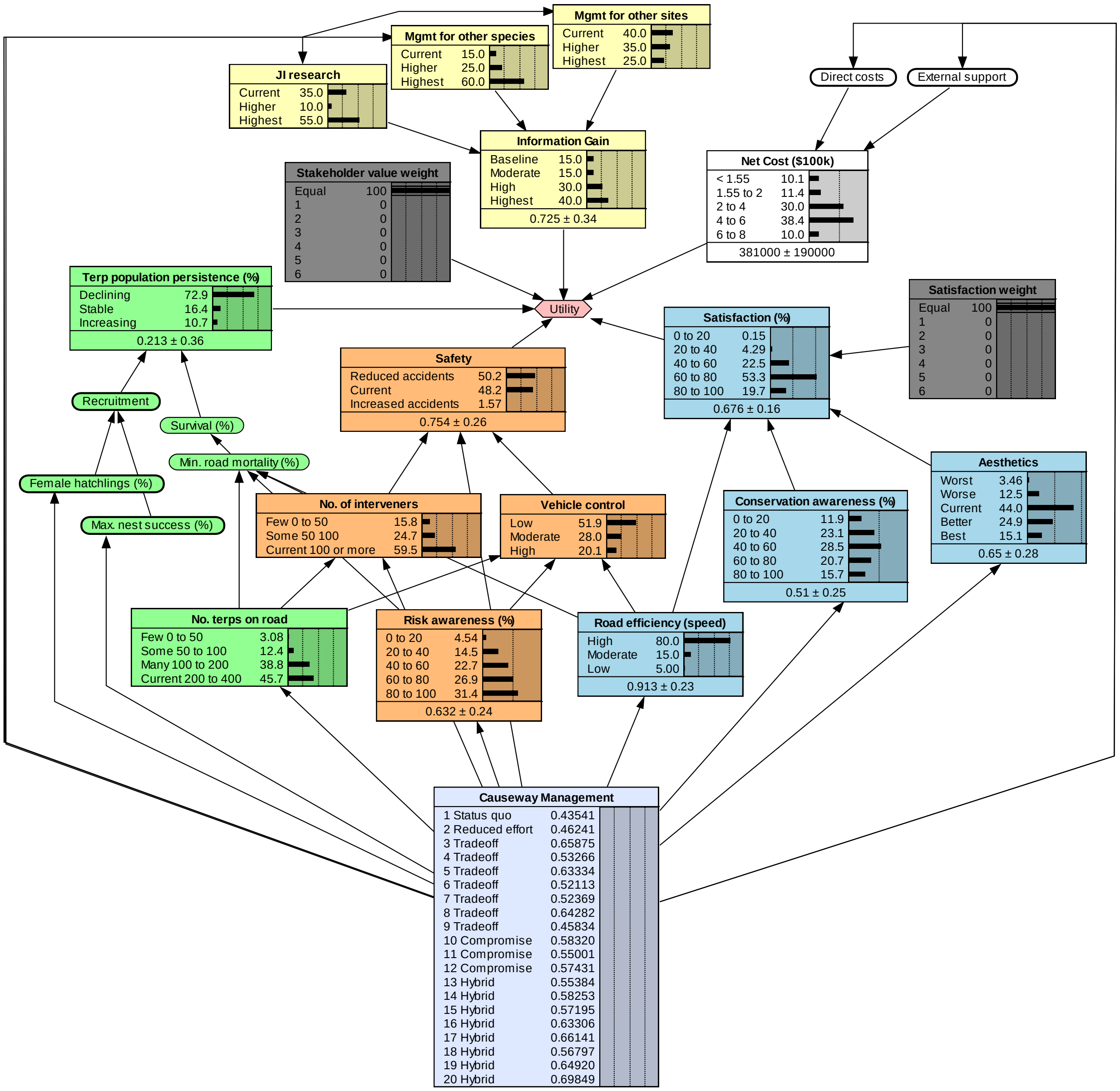


Figure 5.2. Bayesian belief network representing relationships between the five fundamental objectives, means objectives, and management strategies for the Jekyll Island Causeway, Jekyll Island, GA. Colors indicate different fundamental objectives and their means objectives. Grey nodes represent modeling of different stakeholder weights for fundamental and satisfaction objectives, and are pictured as set to “Equal” representing the main analysis that used mean objective weights across all stakeholders.

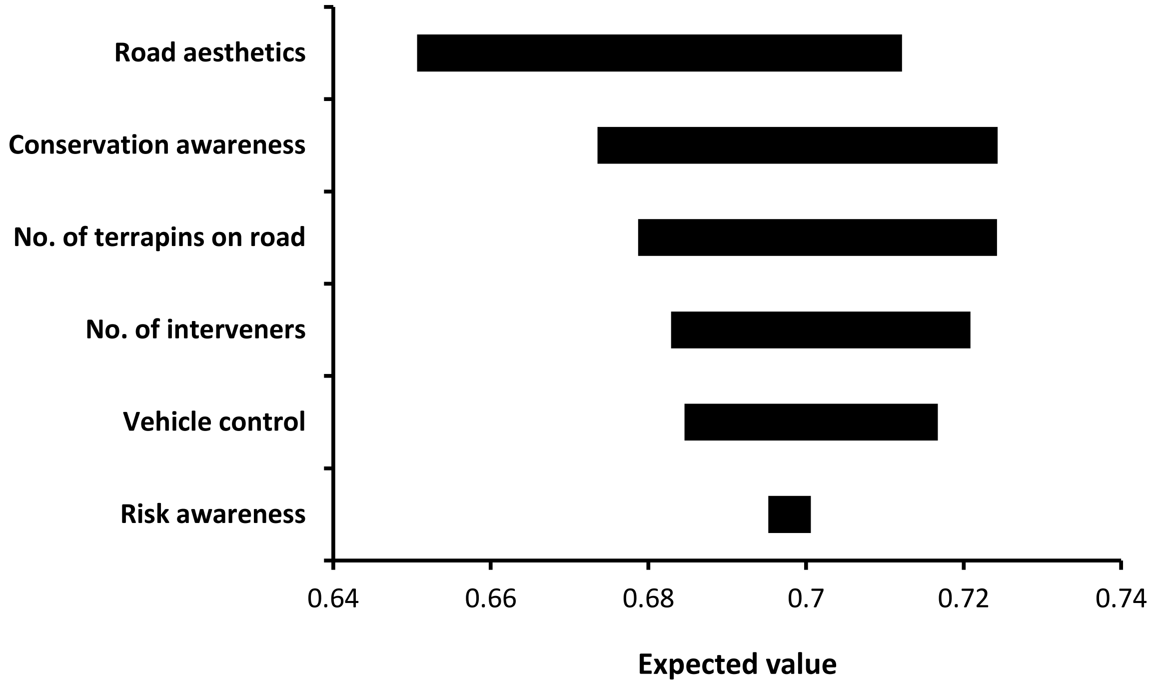


Figure 5.3. Tornado diagram for one-way sensitivity analysis of decision model components influencing safety and satisfaction objectives. Components are arranged from most (top) to least (bottom) influential for the Jekyll Island Causeway management decision.

CHAPTER 6

APPLYING AN INTEGRATIVE MODEL OF BEHAVIOR CHANGE TO IMPROVE GLOBAL CONSERVATION OUTCOMES[4]

---

[4] Crawford, B.A., K.M. Green, and A.A. DeWan, To be submitted to *Conservation Letters*

ABSTRACT

The rapidly increasing rate of biodiversity and habitat loss across the globe is largely the result of destructive human behaviors. Although many conservation initiatives have addressed human behavior-driven threats, particularly through education, little evidence supports their success. We evaluated the impact of 84 projects that moved beyond traditional awareness-raising and applied the same integrated model of behavior change and associated interventions to target destructive activities such as illegal hunting, overfishing, and agricultural practices. Collectively the projects conducted pre/post intervention sociological surveys of more than 20,000 individuals in 18 countries to measure changes in behavioral model variables including: knowledge, attitudes, interpersonal communication, behavior intention, and behavior. We assessed the efficacy of behavior change interventions across sites and tested 7 behavior change models to determine the relative importance and order in which changes in each variable influence changes in behavior. Across interventions, all behavior change model variables demonstrated significant increases ($p < 0.001$) ranging from 16.1 - 25.0 percentage points. The fully integrated behavior change model (all variables included) was the most parsimonious and explained 71% of the observed variance in behavior change. Behavior intention moderated the influence of all other variables on behavior change, had the largest effect of any predictor variable in the model. Based on an unprecedented sample of systematic interventions, our results highlight the importance of incorporating behavioral theory into traditional conservation programs and the potential value of applying this new integrated behavior change model to address threats to biodiversity across the globe.

INTRODUCTION

Recent studies predict that current species extinction rates are 100 to 1,000 times pre-human background rates (Pimm *et al.* 1995; Millennium Ecosystem Assessment 2005) and some species have lost over 50 percent of their historic range (Ceballos & Ehrlich 2002). Threats such as destructive fishing, illegal timber harvesting, ivory poaching, and bushmeat hunting are at the core of species losses across the globe (Ceballos & Ehrlich 2002; Schultz 2011). Most experts now agree that these worsening environmental conditions are largely the result of human behaviors (e.g., St John *et al.* 2010; Schultz 2011) and that achieving conservation outcomes is fundamentally about changing behaviors (Ehrlich & Kennedy 2005). Although awareness-raising initiatives have been a common tool applied to address human behavior-driven threats, education alone rarely translates into behavior change (Crohn & Birnbaum 2010; McKenzie-Mohr 2011). The field of public health has recognized the need to move beyond traditional awareness-raising, applying targeted campaigns grounded in theory from the behavioral sciences (Evans *et al.* 2014). Conservation practitioners have made less progress in this direction, but calls for integrating behavior change principles into traditional conservation programs are appearing more often in the literature (e.g., St John *et al.* 2010; Schultz 2011; Mastrangelo *et al.* 2014). Some have even said that the field of conservation science "has largely failed to embrace the notion that the study of human choice about nature conservation is potentially the most important research topic in the world of today" (Cowling 2014).

Human behavioral sciences offer critical insights for more effectively integrating the role of people into conservation solutions (Heimlich & Ardoin 2008). Behavior change research, in particular, explains how and why humans make choices that may negatively impact themselves, their community and their environment. Numerous theoretical models explain why knowledge and awareness do not necessarily lead to pro-environmental outcomes, even when those outcomes are objectively beneficial to the individual (Hines *et al.*1987). Social-psychological models such as the transtheoretical model of behavior change (Prochaska 1979) and the theory of planned behavior (Ajzen 1991) are among the most applied and tested approaches for describing why and how humans make decisions about how to act and when to change behaviors.

The transtheoretical model of behavior change (Prochaska 1979) suggests that individuals move through a series of cognitive stages in the transition to any new or different behavior. Increasing awareness and changing attitudes lead to an increased willingness and preparation before actually performing and maintaining the new behavior (Prochaska 1979). The occurrence of these stages has been generalized and documented across numerous problem behaviors, particularly in the health sector (Prochaska *et al.* 1999). Further applications of this model have been influenced by diffusion of innovation theory (Rogers 2010), predicting higher rates of behavior change when people are engaged in communication about the behavior (Vaughan & Rogers 2000) and when they observe its performance by others in the same social group (Bandura 1986).

The theory of planned behavior (Ajzen 1991) posits a different model, emphasizing the importance of perceived behavioral control, derived from self-efficacy theory (Bandura 1977). This differs from the transtheoretical model in that changing attitudes does not influence behavior directly. Instead attitudes influence behavioral intentions, which in turn shape our actions (Bamberg & Möser 2007). This model is considered the most influential attitude-behavior model in social psychology (Kollmuss & Agyeman 2002), where knowledge of the issues and solutions, an individual’s perception of control to bring about change, attitudes, and verbal commitment were identified as critical prerequisites for pro-environmental behavior

(Hines *et al.* 1987). Although there are a few applications of this approach in a conservation setting (see Mastrangelo *et al.* 2014), there has been increased demand to apply these models to behaviors that have a large impact on biodiversity (Gardner & Stern 2002; St John *et al.* 2010).

Both the Prochaska (1979) and Ajzen (1991) models, among others, have been adapted to increase explanatory power in different contexts (e.g., Mastrangelo *et al.* 2014). These models, rather than being irreconcilably distinct, offer key insights that can be integrated, applied and tested to improve the practice of conservation programs. Here we build from previous work to develop and test an integrative model of behavior change for conservation (Figure 6.1).

Our model refines the knowledge, attitudes, and interpersonal communication factors identified in previous approaches into distinct constructs that have been shown to have varying influences on behavior change in other models (Hines *et al.* 1987). For example, systems knowledge (i.e., knowledge of the issue itself) and solutions knowledge (i.e., knowledge of the strategy to solve the issue) have each been found to influence attitudes and behaviors (Frick *et al.* 2004). Similarly, although attitudes generally reflect one’s beliefs about the benefits and consequences of a behavior (Kollmuss & Agyeman 2002), they can be further disaggregated to include two additional constructs: normative attitudes and barrier attitudes. The former reflect one’s beliefs about what is expected or acceptable among a larger community of peers to which the person belongs, while the latter denote feelings about obstacles to a behavior that are either intrinsic (i.e., the ease of performing the new behavior) or extrinsic (i.e., the resources and infrastructure available in the community to facilitate behavior adoption and effectiveness). Finally, we acknowledge the importance of behavior intention as a mediator that is influenced by various types of knowledge and attitudes and, in turn, has the strongest direct influence on behavior change (Bamberg & Möser 2007). Collectively, these modifications form the foundation for the new integrative model.

In this study, we conducted a meta-analysis of 84 targeted behavior change interventions designed to drive behavior change in a variety of social and ecological contexts. Each intervention applied the integrative model of behavior change (Figure 6.1) in order to replace destructive natural resource practices with more sustainable behaviors. This is the first study of its kind and scale to comprehensively apply and evaluate the potential of targeted behavior change interventions to tackle complex biodiversity conservation challenges across the globe. Our results demonstrate that integrating behavioral theory into traditional interventions can be a powerful tool for changing behaviors and improving global conservation outcomes.

METHODS

We evaluated data from 84 targeted behavior change interventions at sites in 18 countries between 2009 and 2012. These community-level interventions were systematically implemented by the non-profit organization Rare (Jenks *et al.* 2010) and its non-governmental and local government agency partners. Each intervention employed standardized sociological surveys pre- and post-intervention tailored to the conservation, social, or behavioral context at the site. Using a meta-analytic approach, we 1) extracted and validated data for variables associated with the key constructs of the model, 2) estimated mean effect sizes for each variable across all interventions, and 3) used path analysis to measure relationships among variables included in the full integrated behavior change model and alternative models.

Data collection and coding

We compiled a database from survey data obtained before and after interventions measuring changes in community knowledge, attitudes, interpersonal communication, and behaviors (see DeWan *et al.* 2013; Green *et al.* 2013). It is important to note that all interventions collected self-reported behavior change data via survey methodologies and did not directly measure observed behaviors. Although some studies have found discrepancies between self-reported and observed environmental behaviors, especially when the behavior is socially desirable (Corral-Verdugo 1997), most researchers continue to rely on self-reported data in the absence of reliable, cost-effective techniques to measure actual behavior frequencies (Steg & Vlek 2009).

On-site researchers converted primary survey responses (originally on “yes”/ “no” or “agree”/ “disagree” scales) into binary data where responses indicating the targeted outcomes of interventions (e.g., respondent was aware of conservation solutions, agreed with efficacy of proposed behavior solution, or reported behavior change) = 1 and neutral or negative responses = 0. Thus, all data collected for the meta-analysis included the number of people in each response category and the sample size for pre- and post-intervention surveys. We reviewed all original survey questions and assigned corresponding data to one of eight variables associated with our model (Table 6.1); all variables were mutually-exclusive and grounded in behavioral theory. Most survey questions distinctly measured one variable in the model, which was determined by coding agreement between researchers. When a group of researchers deemed a question ambiguous, it was not included in the analysis; therefore, we did not extract data for each variable from each intervention (see Table 6.2). These procedures, typical of meta-analyses (e.g., Bamberg & Möser 2007), ensured the consistency of constructs measured across independent studies and the validity of pooling these results. During this stage, we confirmed data came from a primary audience directly involved in the conservation issue (e.g., fishers) and that a causal link between the conservation issue and targeted behavior existed between all survey questions measuring each variable within an intervention, from systems and solutions knowledge to behavior change, that were included in the analysis.

Effect size analysis

We calculated mean effect sizes for interventions as the percentage point (pp) difference (post – pre) in community members exhibiting pro-environmental states of each variable (e.g., knowing about a behavior solution, believing in its benefits, adopting behavior). We used random-effects meta-regression models with the DerSimonian–Laird method within the ‘metafor’ package (Viechtbauer 2002) in R (R Core Team 2013), which do not assume one true effect and instead estimate the mean effect size from a distribution of effects that can vary across studies (Borenstein *et al.* 2011). We suspected substantial heterogeneity to exist between interventions in our dataset (which was later confirmed, see Results) due to variation in specific implementation methods, variation in the new behavior’s degree of difficulty, and community variation in geography, demography, and culture that we have not or could not capture completely by including additional factors in our models. This procedure pooled effect sizes of each variable using inverse variance weighting that incorporates within and between-intervention variance, which gives more weight to effects derived from interventions with larger sample sizes (Borenstein *et al.* 2011). Since we had access to all intervention data regardless of its use in publications or other reports, we did not perform any tests for publication bias frequently used by other meta-analyses (Bamberg & Möser 2007).

Path analysis

To test relationships among variables in our behavior change model, we specified and analyzed several path models including a full integrated model containing all eight variables and several, reduced-parameter alternatives in EQS 6.2 (Bentler 2000). Meta-analyses employing path models (i.e. meta-analytic structural equation modeling: MASEM) typically collect correlations between variables from primary studies, pool correlations into a combined matrix, and test path models on this matrix (e.g., Bamberg & Möser 2007). We had to adapt these methods due to limitations of the primary data currently available. Data collected from Rare contained metrics suitable for meta-regression analyses, including counts of people responding affirmatively to survey items and total sample sizes; however, correlations between variables were never calculated and reported for any individual intervention. Retroactively obtaining correlations from primary data files was not possible due to time and logistical constraints (e.g., older intervention data is not centrally stored, surveys are in several languages). Therefore, we created a table of effect sizes calculated during meta-regression procedures for each variable measured in each intervention under random-effects assumptions, which used inverse variance weighting. We then used these effect sizes as inputs to calculate correlations between all variable pairs using pairwise deletion. This approach allowed us to generate a pooled correlation matrix to test hypothesized path models using the intervention as the unit of analysis ($N = 84$). For all path models, we used robust maximum likelihood methods to estimate path coefficients, similar to regression weights, which correct for small sample sizes, missing data, and moderate multivariate kurtosis (Byrne 2000) that were present in our dataset, and we report resulting robust statistics. Goodness of fit for all models in our path analysis was evaluated on several criteria. We used the Yuan-Bentler residual χ^2 that provides more reliable test statistics given smaller sample sizes and mild nonnormality of data (Bentler & Yuan 1999), where nonsignificant results indicate the model is consistent with the observed data. Bentler's comparative fit index (CFI) and root-mean-square error of approximation (RMSEA) were also used to examine model fit (Bentler 2000). Although several recommendations exist for setting index cutoff values, it is generally accepted that values of CFI < 0.90 and RMSEA > 0.10 indicate poor model fit, whereas larger CFI (0.95 – 1.0) and smaller RMSEA (0.0 – 0.06) are optimal criteria for acceptable models, especially for large sample sizes (Cheung & Chan 2005). Lastly, we compared the fit of models relative to each other while accounting for parsimony using Akaike's Information Criteria (AIC), where lower values indicate better fit relative to other candidate models (Akaike 1974).

We followed these methods to test relationships among variables leading to behavior change in three sets of path models. First, we tested an environmental education model to serve as a baseline for comparison by specifying a path model where only systems and solutions knowledge were included as direct predictors of behavior change. Second, we specified a set of four alternative models to determine the best-supported relationships between interpersonal communication with knowledge and attitudes (IC1-4), given that communication has been proposed as an important antecedent to behavior change but has not yet been tested (Vaughan & Rogers 2000). For each model in this set, all three attitude variables were hypothesized to mediate the influence of knowledge types on behavior change (Hines *et al.* 1987; Frick *et al.* 2004; Bamberg & Möser 2007) but models differed in treating interpersonal communication as a predictor or response variable relative to knowledge and/or attitudes. In the last set of models, we created a full integrated model using the best-performing relationships between knowledge, attitudes, and interpersonal communication found in IC1-4. The full model included all eight variables measured in our study. We tested a final model in this set by removing nonsignificant paths found in the full integrated model in order to improve goodness of fit (Figure 6.2).

RESULTS

Effect size analysis

From each of the 84 interventions, we extracted data for three to all eight variables (median = 5) defined in the full integrated behavior change model. Mean sample size for intervention surveys measuring variables was 261 respondents from the primary audience (range = 31 to 1192). Between-intervention heterogeneity in effect sizes were statistically significant for all variables ($Q_{29\text{-}78}$ ranged from 1102 to 19496, all $P < 0.001$), which justified the use of random-effects models. Across interventions, mean effect sizes were significant and positive for all behavior change model variables ($P < 0.001$) ranging from 16.1 - 25.0 percentage points, including an 18.1 percentage point increase in behavior change (Table 6.2).

Path analysis

Table 6.3 shows information for pooled correlations among variables measured in this study. The upper triangular matrix presents the number of interventions (out of 84 included in the meta-analysis) that measured each variable pair, which ranged from 12 (barrier removal attitude and behavior intention) to 71 (interpersonal communication and behavior change). Although most interventions did not measure all eight variables, we were able to extract an adequate number of effects from interventions necessary to calculate all bivariate correlations. The lower triangular matrix presents estimated confidence intervals for pooled correlations resulting from intervention effect sizes calculated under the random-effects assumption. The 95% confidence intervals around mean correlation coefficients were positive and did not include 0 for most variable pairs (indicating significance at the $P < 0.05$ level). Variables that did not correlate significantly were systems knowledge with normative attitudes, solutions knowledge with normative attitudes, and barrier removal attitudes with solutions knowledge, normative attitudes, interpersonal communication, and behavior intention.

The full integrated behavior change model (Figure 6.2) performed best given goodness of fit criteria and a ΔAIC value > 2 of the next best model (Table 6.4) and explained 71% of the observed variance in behavior change. All path coefficients were significant ($P < 0.05$) and positive in sign for the full integrated model, indicating direct relationships between the effects of interventions on predictor and response variables in behavior change models. Behavior intention moderated the influence of all other variables on behavior change and had the largest effect of any predictor variable in the model. The influences of benefits and normative attitudes on behavior intention were relatively equal in magnitude, and interpersonal communication and systems, but not solutions, knowledge influenced each of the three attitude types. The environmental education model performed poorly in all criteria measuring goodness of fit and significantly departed from the observed data. Among the interpersonal communication model set (IC1-4), IC4, where interpersonal communication and knowledge covaried while directly influencing attitude variables, was the best-performing model that met all goodness of fit criteria, and these relationships were used in the full integrated model.

DISCUSSION

Our study employed a uniquely large sample of systematic interventions to evaluate the efficacy of integrating behavior change theory into biodiversity conservation strategies. Although some recent strategies have sought to incorporate behavioral thinking (see, for example, Ardoin *et al.* 2013), our study used a comprehensive data set to test and strengthen the power of using these types of tools in addressing human-driven threats to biodiversity. Our results build from previous social-psychological models that have been predominantly applied in public health to posit an integrative behavior change model that explains the drivers of behavior change in some of the world's most threatened ecosystems. The lessons described here can be rapidly applied to existing and future programs to better address the behavioral precursors that underlie conservation problems.

Our results demonstrated large overall changes in behavior across interventions (95% confidence interval: 12.1 – 24.0 pp), providing new evidence that interventions that apply this model can be effective for addressing challenging and widespread conservation problems. These changes exceeded those found in other studies of behavior change interventions for public health and safety concerns, including smoking, cancer screening and seat belt usage (Snyder *et al.* 2004). These changes also ranked among the highest observed for environmental or conservation interventions (Schultz 1999) over a short period of time; furthermore, interventions conducted in our study targeted behaviors that were likely more difficult and personally-costly (e.g., adhering to stricter fishing regulations, participating in community reciprocal watershed agreements) relative to those interventions (e.g., recycling, reducing littering or electricity usage). It is important to note, however, that the duration of behavior change interventions reflected in this study lasted less than 3 years. As De Young (1991) suggests, measures of success for behavior change interventions will ultimately need to be reflected in the persistence of these changes over time, without repeated intervention. Furthermore, the underlying purpose of behavior change interventions is to reduce human impacts on biodiversity and contribute to sustainable wildlife populations (Jenks *et al.* 2010). Although these data are not currently available, future efforts should evaluate the persistence of documented behavior changes over time and the long-term effects these changes have on biodiversity outcomes.

Path analysis offered a useful tool for testing the relationships among traditional behavior change variables as well as comparing the ability of the full integrated model to predict conservation behavior change relative to simpler, alternative models. Our full integrated model was the best predictor of behavior changes measured in interventions with an explained observed variation of 71%. In a complementary analysis (not presented here), removing each behavior change antecedent, in turn, from the full integrated model resulted in poor model fit, indicating all were necessary to accurately predict a community's degree of behavior change. Multiple components and relationships of the best-supported model agree with previous behavior change studies. Similar to relationships postulated in the theory of planned behavior (Ajzen 1991), we found evidence that a community's norms and attitudes significantly predicted behavior intention, which, in turn, had the greatest influence on behavior change among all other model variables. Our results also support findings of Bamberg & Möser (2007) that awareness of threats and behavioral solutions is a necessary precursor to a community's attitudes and subsequent behaviors.

In contrast to the full integrated model, the traditional environmental education model (in which knowledge alone leads to behavior change) was one of the lowest ranked of those tested. These results echo previous findings that increasing environmental knowledge is necessary but insufficient by itself to facilitate pro-environmental behaviors (Blumstein & Saylan 2007; McKenzie-Mohr 2011). Thus, basic environmental awareness and education programs can be strengthened by including more comprehensive efforts to target changes in a community's values, norms, communication, and willingness to try a new behavior (Blumstein & Saylan 2007; Heimlich & Ardoin 2008). Such efforts may draw on the approach used by Rare, and the broader field of community-based social marketing, designed to promote psychological, and not only cognitive, components of a new behavior beneficial to the community and environment (Schultz 2014).

At the interplay between cognitive and psychological components, our results offered some of the first empirical evidence for the role interpersonal communication plays in facilitating behavior change. In the full integrated model, changes in knowledge and interpersonal communication were reinforcing and provide a critical pre-condition to changes in attitudes (Figure 6.2). These hypothesized relationships (also included in IC4) were based on plausibility that communication about a new behavior between two people required some prior awareness but could also increase each person's knowledge of the subject; alternatively, a particular attitude is not a prerequisite for communication, but discussing behaviors with peers will likely influence attitudes. This configuration highlights the important role of community conversations to share, refine, or reinforce knowledge while simultaneously nurturing attitudes targeted by conservation interventions. Although the relationships between communication and other behavior antecedents had not been previously characterized, social marketing interventions, such as those assessed here, focus strongly on promoting channels of communication in addition to removing any barriers (psychological or structural factors that make behavioral adoption more difficult: Schultz 2014). Mean intervention effect sizes were highest for interpersonal communication, indicating that interventions were especially effective at promoting peer-to-peer discussions about local conservation issues and behavioral solutions. This is likely at least in part due to low baseline conditions where conversations about the problem or the solution were less common and activities are specifically tailored to increase these types of discussions.

The strategic intervention framework used in these projects goes beyond previous efforts to induce changes in behavior, to demonstrate some of the highest observed changes for environmental or conservation interventions (Schultz 1999). These approaches may be more successful than others because of the intense focus on driving interpersonal communication through public events and community engagement or because they are being applied to smaller, close-knit communities where social norms play a stronger role in defining individual behavior (Jenks *et al.* 2010). This study advances previous behavior change models to clearly demonstrate that targeted interventions that capitalize on changing attitudes and conversations in key biodiversity areas offer a unique solution to some of the world's most challenging conservation problems.

ACKNOWLEDGEMENTS

We thank J. Whitney, C. Austin, K. Schwartz, E. Tsibulevskiy, and M. Pearce for support in data gathering, compilation and preparation and S. Gaines, E. Soule, and P. Ferraro for comments and discussions on the study. We thank J. C. Maerz and the Center for Integrative Conservation Research (University of Georgia) for fellowship coordination and support for B.C. We thank Rare's many partners in the field for their efforts to apply the behavior change model and to collect the data that has enabled this study. Finally, we thank P. Butler and B. Jenks for their leadership in supporting the study.

REFERENCES

Ajzen, I. (1991). The theory of planned behavior. *Organ. Behav. Hum. Decis. Process.*, **50**, 179-211.

Ajzen, I. & Fishbein, M. (1980). *Understanding Attitudes and Predicting Social Behaviour*. Prentice Hall, Englewood-Cliffs, NJ.

Akaike, H. (1974). A new look at the statistical model identification. *Automatic Control, IEEE Transactions*, **19**, 716-723.

Ardoin, N. M., Clark, C., & Kelsey, E. (2013). An exploration of future trends in environmental education research. *Environmental Education Research*, **19**, 499-520.

Bamberg, S. & Möser, G. (2007). Twenty years after Hines, Hungerford, and Tomera: A new meta-analysis of psycho-social determinants of pro-environmental behaviour. *J. Env. Psych.*, **27**, 14-25.

Bandura, A. (1977). Self-efficacy: Toward a unifying theory of behavioral change. *Psychol. Rev.*, **84**, 191.

Bandura, A. (1986). *Social Foundations of Thought and Action: A Social Cognitive Theory*. Prentice Hall, Englewood-Cliffs, NJ.

Bentler, P.M. (2000). *EQS 6 Structural Equations Program Manual*. Multivariate Software, Inc., Encino, CA.

Bentler, P.M. & Yuan, K.H. (1999). Structural equation modeling with small samples: Test statistics. *Multivariate Beh. Research*, **34**, 181-197.

Blumstein, D.T. & Saylan, C. (2007). The failure of environmental education (and how we can fix it). *PloS Bio.*, **5**, 973-977.

Borenstein, M., Hedges, L.V., Higgins, J.P. & Rothstein, H.R. (2011). *Introduction to meta-analysis*. Wiley, Chichester, UK.

Byrne, B.M. (2001). Structural equation modeling with AMOS, EQS, and LISREL: Comparative approaches to testing for the factorial validity of a measuring instrument. *Int. J. Testing*, **1**, 55-86.

Ceballos, G. & Ehrlich, P.R. (2002). Mammal population losses and the extinction crisis. *Science*, **296,** 904-907.

Cheung, M.W.L. & Chan, W. (2005). Meta-analytic structural equation modeling: a two-stage approach. *Psych. Methods*, **10**, 40-64.

Corral-Verdugo, V. (1997). Dual 'realities' of conservation behavior: Self-reports vs observations of re-use and recycling behavior. *J. Environ. Psychol.*, **17**, 135-145.

Cowling, R.M. (2014). Let's Get Serious About Human Behavior and Conservation. *Conservation Letters*, **7**, 147-148.

Crohn, K. & Birnbaum, M. (2010). Environmental education evaluation. *Eval. And Prog. Planning*, **33,** 155-158.

De Young, R. (1993). Changing behavior and making it stick: The conceptualization and management of conservation behavior. *Environment and Behavior*, **25**, 485-505.

DeWan, A., Green, K., Li, X. & Hayden, D. (2013). Using social marketing tools to increase fuel-efficient stove adoption for conservation of the golden snub-nosed monkey, Gansu Province, China. *Cons. Evidence*, **10**, 32-36.

Ehrlich, P. & Kennedy, D. (2005). Millennium assessment of human behavior. *Science*, **309,** 562-563.

Evans, W.D., Pattanayak, S.K., Young, S., Buszin, J., Rai, S. & Bihm, J.W. (2014). Social marketing of water and sanitation products: A systematic review of peer-reviewed literature. *Soc. Sci. & Med.*, **110**, 18-25.

Frick, J., Kaiser, F. & Wilson, M. (2004). Environmental knowledge and conservation behavior: Exploring prevalence and structure in a representative sample. *Pers. Indiv. Diff.*, **37,** 1597-1613.

Gardner, G.T. & Stern, P.C. (1996). *Environmental problems and human behavior*. Allyn & Bacon, Needham Heights, MA.

Green, K.M., DeWan, A., Arias, A.B. & Hayden, D. (2013). Driving adoption of payments for ecosystem services through social marketing, Veracruz, Mexico. *Cons. Evidence*, **10**, 48-52.

Heimlich, J.E. & Ardoin, N.M. (2008). Understanding behavior to understand behavior change: a literature review. *Env. Edu. Res.*, **14**, 215-237.

Hines, J.M., Hungerford, H.R. & Tomera, A.N. (1987). Analysis and synthesis of research on responsible environmental behavior: A meta-analysis. *J. Environ. Educ.*, **18**, 1-8.

Hu, L.T. & Bentler, P.M. (1999). Cutoff criteria for fit indexes in covariance structure analysis: Conventional criteria versus new alternatives. Structural Equation Modeling: A Multidisciplinary Journal, **6**, 1-55.

Jenks, B., Vaughan, P. & Butler, P. (2010). The evolution of Rare Pride: Using evaluation to drive adaptive management in a biodiversity conservation organization. *Eval. Program Plann.*, **33,** 186-190.

Kollmuss, A. & Agyeman, J. (2002). Mind the gap: Why do people act environmentally and what are the barriers to pro-environmental behavior? *Environ. Educ. Res.*, **8**, 239-260.

Mastrangelo, M.E., Gavin, M.C., Laterra, P., Linklater, W.L. & Milfont, T.L. (2014). Psycho-social factors influencing forest conservation intentions on the agricultural frontier. *Conservation Letters,* **7**, 103-110.

McKenzie-Mohr, D. (2011). *Fostering sustainable behavior: An introduction to community-based social marketing*. 3rd ed. New Society Press, Gabriola Island, Canada.

Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being*. Island Press, Washington, D.C.

Pimm, S.L., Russell, G.J., Gittleman, J.L. & Brooks, T.M. (1995). The future of biodiversity. *Science*, **269,** 347-350.

Prochaska, J. (1979). *Systems of psychotherapy: A transtheoretical analysis*. Dorsey Press, Belmont, CA.

Prochaska, J.O., DiClemente, C.C. & Norcross, J.C. (1999). In search of the structure of change. Pages 87-114 in Y. Klar, J.D. Fisher, J.M. Chinsky & A. Nadler, editors. *Self Change: Social Psychological and Clinical Perspectives.* Springer, New York, NY.

R Core Team. (2013). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. Retrieved from http://www.R-project.org/.

Rogers, E. (2010). *Diffusion of Innovations*. 4th ed. The Free Press, New York, NY.

Schultz, P. (1999). Changing behavior with normative feedback interventions: A field experiment on curbside recycling. *Basic Appl. Soc. Psych.,* **21**, 25-36.

Schultz, P.W. (2011). Conservation means behavior. *Conserv. Biol.,* **25**, 1080-1083.

Schultz, P.W. (2014). Strategies for promoting proenvironmental behavior: Lots of tools but few instructions. *European Psychologist,* **19**, 107-117.

Snyder, L.B., Hamilton, M.A., Mitchell, E.W., Kiwanuka-Tondo, J., Fleming-Milici, F. & Proctor, D. (2004). A meta-analysis of the effect of mediated health communication campaigns on behavior change in the United States. *J. Health Comm.*, **9,** 71-96.

St John, F., Edwards-Jones, G. & Jones, J. (2010). Conservation and human behaviour: Lessons from social psychology. *Wildlife Res.*, **37,** 658-667.

Steg, L. & Vlek, C. (2009). Encouraging pro-environmental behaviour: An integrative review and research agenda. *J. Environ. Psychol.*, **29,** 309-317.

Vaughan, P. & Rogers, E. (2000). A staged model of communication effects: Evidence from an entertainment–education radio soap opera in Tanzania. *J. Health Commun*., **5**, 223-227.

Viechtbauer, W. (2010). Conducting meta-analyses in R with the metafor package. *J. Stat. Software*, **36**, 1-48.

Table 6.1. Definitions and representative examples of behavioral model variables.

Variable	**Definition**	**Representative questionnaire item**
Knowledge		
Systems	Cognition of environmental objects (e.g., species, habitats) or threats (e.g., overfishing)	Deforestation in the highlands decreases water availability in the lower basin (true/false)
Solutions	Cognition of strategies designed to reduce environmental threats	Upstream landholders can receive incentives for not cutting down the forest on their properties (true/false)
Attitudes		
Barrier Removal	Beliefs about the presence and efficacy of skills, resources, or infrastructures needed to implement the conservation strategy	The community has adequate infrastructure, equipment, and facilities to enforce the rules of the no-take zone (agree/disagree)
Benefits	Feelings regarding the benefits of the conservation strategy	The implementation of the co-management strategy will improve fishing production in the Hanjiang River (agree/disagree)
Normative	Beliefs about the moral obligations of a person and others to perform the conservation behavior	The Arbor-Granada Marine Sanctuary regulations need to be followed by all people (agree/disagree)
Interpersonal communication	Statements about communicating and validating information among community members regarding current human impacts and the conservation behavior	Have you talked with other villagers about joining the community co-management committee during the past six months? (yes/no)
Intention	Statements about the intention to adopt the conservation behavior	How willing are you to implement land conservation mechanisms on your farm? (willing/unwilling)
Behavior	Conservation behavior is adopted by person	In the last 6 months, have you released gravid females and under-sized lobsters during live lobster fishing? (yes/no)

Table 6.2. Overall summary effects of interventions on variables in an integrated behavior change model, as estimated by random-effects meta-regression.

	Sample Size Information			**% point change (post - pre)**	
Model Variable	***K***[*]	***N***[†] **Pre**	***N***[†] **Post**	**Estimate**	**95% CI**
Knowledge					
Systems	53	15,617	14,781	18.1	12.8, 23.5
Solutions	73	18,228	17,415	19.9	8.8, 30.9
Attitudes					
Benefits	51	14,401	13,543	19.3	14.1, 24.4
Normative	48	13,550	12,552	17.9	13.1, 22.8
Barrier Removal	30	8,186	7,977	16.1	8.4, 23.9
Interpersonal communication	79	19,109	18,165	25	18.2, 31.8
Intention	48	14,318	13,420	16.2	7.8, 24.58
Behavior	76	19,485	18,383	18.1	12.1, 24.0

[*] Number of interventions

[†] Number of community members sampled

Table 6.3. 95% confidence intervals (lower triangular matrix) of pooled correlations and number of interventions (upper triangular matrix) that measured each pair of variables. Intervals in bold indicate significant correlations ($P < 0.05$).

Variable	1	2	3	4	5	6	7	8
1. Knowledge - systems	53	44	46	19	34	50	36	48
	-							
2. Knowledge - solutions	**0.17**	73	43	27	41	67	40	64
	0.65	-						
3. Attitudes - barrier removal	**0.07**	-0.04	30	19	15	27	12	26
	0.78	0.64	-					
4. Attitudes - benefits	**0.36**	**0.12**	**0.02**	51	31	48	32	44
	0.78	**0.63**	**0.76**	-				
5. Attitudes - norms	-0.02	-0.12	-0.10	**0.38**	48	45	25	43
	0.59	0.47	0.77	**0.81**	-			
6. Interpersonal comm.	**0.05**	**0.47**	-0.03	**0.34**	**0.13**	79	44	71
	0.55	**0.76**	0.65	**0.74**	**0.63**	-		
7. Intention	**0.44**	**0.19**	-0.21	**0.56**	**0.40**	**0.31**	48	42
	0.82	**0.68**	0.80	**0.88**	**0.85**	**0.73**	-	
8. Behavior change	**0.40**	**0.01**	**0.02**	**0.47**	**0.33**	**0.18**	**0.62**	76
	0.76	**0.47**	**0.68**	**0.80**	**0.74**	**0.57**	**0.87**	-

Table 6.4. Goodness of fit results for the full integrated behavior change path model (top two models) and alternative models.

Model Name	**Description**	χ^2	**df**	**p-value**[a]	**CFI**[†]	**RMSEA**[‡]	**AIC**[§]
Full integrated model (trimmed)	Nonsignificant paths removed	16.2	9	0.440	0.95	0.08	-7.999
Full integrated model	Includes all behavior antecedents in study	10.8	12	0.544	0.94	0.10	-1.707
IC4	IC[¶]/knowledge affect attitudes	8.2	6	0.220	0.94	0.10	-1.040
IC3	IC affects knowledge/attitudes	9.8	7	0.201	0.92	0.11	-0.645
IC1	IC removed	11.0	5	0.051	0.83	0.15	3.813
IC2	Knowledge/attitudes affect IC	11.0	6	0.088	0.87	0.15	4.435
Environmental Education	Knowledge affects behavior change	4.2	1	0.040	0.72	0.29	5.904

[*] Nonsignificant values indicate adequate fit between models and observed data[27]

[†] Bentler's comparative fit index; values of 0.9 or above indicate adequate fit[23]

[‡] Root-mean-square error of approximation; values of 0.1 or below indicate adequate fit[28]

[§] Akaike's Information Criteria; lower values indicate better model fit while accounting for parsimony[29]

[¶] Interpersonal communication

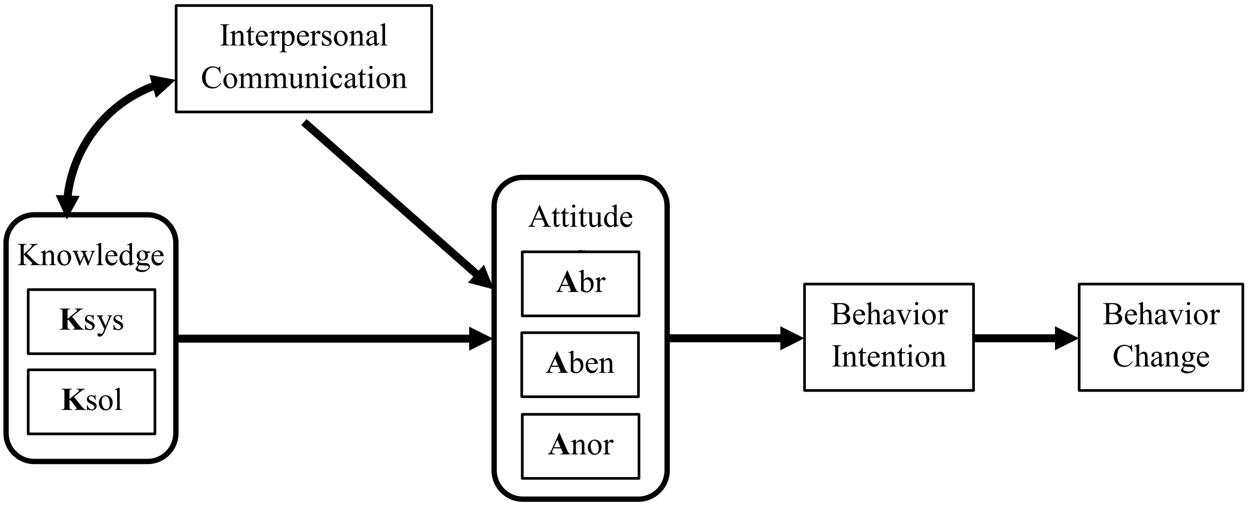


Figure 6.1. Integrative model of behavior change. Hypothesized model of behavior change where changes in systems and solutions knowledge fostered by interpersonal communication can lead to shifts in attitudes and behavior intention that ultimately support changes in behavior. Ksys, systems knowledge; Ksol, solutions knowledge, Abr, barrier removal attitudes; Aben, benefits attitudes; Anor, normative attitudes (see Table 6.1 for examples).

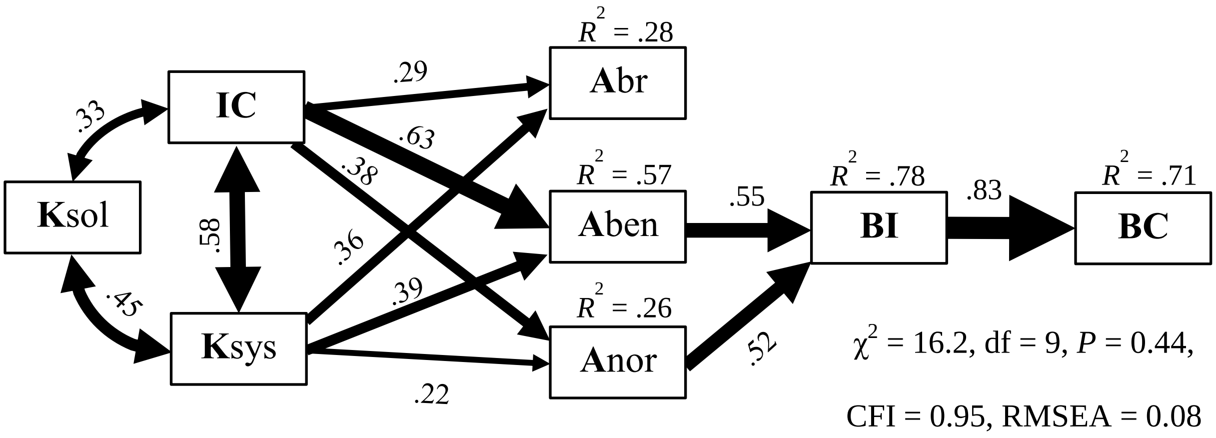


Figure 6.2. Path analysis of integrative model of behavior change. Effects between variables in the best-performing full integrated model of behavior change with nonsignificant paths (*P* > 0.05) removed. Numbers adjacent to single-headed arrows are standardized path coefficients, expressing the direction and strength of relationships between variables, similar to regression weights. Numbers next to double-headed arrows are correlations, expressing reciprocal relationships specified in the model. Arrow width is proportional to the relationship's strength. The proportion of explained variance (R^2) is given for all variables with antecedents. Goodness of fit statistics are shown in the lower right corner where nonsignificant results indicate the model is consistent with the observed data (df, degrees of freedom; CFI, comparative fit index; RMSEA, root-mean-square error of approximation). Model fit and selection results are presented for the full set of tested models in Supplementary Table 6.3. Ksys, systems knowledge; Ksol, solutions knowledge, Abr, barrier removal attitudes; Aben, benefits attitudes; Anor, normative attitudes; IC, interpersonal communication; BI, behavior intention; BC, behavior change.

CHAPTER 7

SYNTHESIS AND CONCLUSIONS

This dissertation has investigated three core themes for developing effective, equitable conservation management solutions: 1) establish a firm understanding of species demography, life history, and behavior, 2) embrace complexity of conservation issues by focusing on multiple threats, actions, stakeholders, and objectives, and 3) integrate humans and their behaviors into solutions instead of viewing them as the source of problems. Across these themes, our work has achieved results that can be applied to conserving diamondback terrapins on Jekyll Island and other sites of high vehicle mortality throughout the species range. Furthermore, our research approaches demonstrate a productive, systematic model for grappling with wicked conservation issues, which can be used to advance conservation management when applied to other human-wildlife conflicts.

ESTABLISH A FIRM UNDERSTANDING OF SPECIES DEMOGRAPHY, LIFE HISTORY, AND BEHAVIOR

We added to the body of knowledge about terrapin demographic rates, life history, and behavior; this information is essential for designing effective management actions and producing models that accurately predict the population’s trajectory. As a species of conservation concern, diamondback terrapins are well-studied in human-impacted landscapes. Still, understanding and estimating population processes remains challenging given cryptic life stages (e.g., hatchlings) that limit rates of detection, dynamic movement through complex habitats that impacts inferences of survival and emigration, and long generation times that require longer study durations to measure population trends.

Certain findings were consistent with previous terrapin studies. In Chapter 2, we characterized nest survival on roadsides and found high levels of predation overall, similar to Roosenburg and Place (1994) and Munscher *et al.* (2012). We also found that nest predation and the proportion of hatchlings born female varied across roadside habitats used for nesting, which allowed us to model changes to these demographic rates given previously untested vegetation management practices (i.e., clearing of roadside hedges). In Chapter 4, we used seven years of intensive mark-recapture and mark-recapture-recover data to corroborate and improve upon demographic estimates from previous terrapin studies. We estimated high survival of females not exposed to road mortality, consistent with survival estimates of populations not under major anthropogenic stressors found by Mitro (2003) and Hart (2005), and higher creek site fidelity for males, relative to females, consistent with findings of Tucker, Gibbons and Greene (2001).

We developed more sophisticated and realistic population models, relative to previous terrapin and road studies, that incorporated key life history and behavioral processes affected by roads. Despite extensive work in the field of road ecology, evaluating demographic and population-level impacts of roads on wildlife has remained challenging. Our work in Chapter 4 is one of the few studies to directly estimate road impacts on survival (Mumme *et al.* 2000; Row, Blouin-Demers & Weatherhead 2007) that advanced previous, limited approaches (e.g., using naïve counts of dead individuals: Aresco 2005; Langen *et al.* 2007). Using mark-recapture-recover and modeling methods (i.e., multistate models) that explicitly accounted for terrapin life history and behavior, we found strong evidence that road mortality was reducing survival of crossing individuals while non-crossers survived at high, natural rates. The use of multistate models to estimate road impacts demonstrates an approach particularly appropriate for other road mortality studies. Multistate models can provide a flexible framework for representing groups differentially impacted by road threats based on behavioral states (i.e., nesting on road vs. otherwise, crossing vs. not crossing). These models also allowed us to separately estimate two fundamental processes of wildlife-vehicle collisions that contribute to the cumulative impact of roads on any local population: the portion of the population exposed to the risk of road mortality and the magnitude of that risk (i.e., the degree to which survival is reduced when crossing). Disentangling these processes was essential in our study, and will likely be for other road studies, when estimating impacts of two types of actions common in road management: barriers that prevent individuals from crossing the road and warning signage that increase survival of crossing individuals by targeting driver awareness. We used robust estimates of demographic rates and effects of management in Chapters 2 and 4 to advance previous terrapin population model efforts (Mitro 2003; Hart 2005; Gilliand, Chambers & LaMar 2014). We expanded simple, stage-based population models (Caswell 1989) to include additional stage- and sex-specific processes relevant to road contexts (e.g., fidelity to roadside nest sites, the proportion of females crossing the road). Ultimately, adapting models to include context-specific threats and demographic rates should produce accurate and reliable population predictions on which management decisions can be based.

The design of certain management actions in Chapters 4 and 5 leveraged information on behavioral patterns of nesting and road crossing to target hot spots and hot moments of mortality (Crawford *et al.* 2014b). Three discrete hot spots of road-crossing activity on the Jekyll Island Causeway (JIC) allowed us to specify and model the implementation of barriers, predator removal, and vegetation management options at these local targets or broader (JIC-wide) scales.

We found that short hybrid barriers, when placed strategically at one hot spot of road crossing, prevented >60% of terrapins in the area from crossing the road. However, management strategies that included barriers, predator removal, and vegetation clearing across broader scales resulted in higher population persistence. Likewise, we used hot moments of activity when implementing flashing warning signage, the first-ever of its kind to target turtles, that flashed around concentrated peaks of nesting (3 hrs per day) associated with the daily diurnal high tide. Static signage has been ineffective at reducing wildlife-vehicle collisions (e.g., Putman 1997) presumably due to driver habituation. Alternatively, signaling drivers within a concentrated period where they are most likely to see a terrapin on the road yielded significant increases in survival of crossing individuals alongside no effect to the survival of non-crossers. The 30% increase in survival associated with flashing signage was higher than any other sign impact study (Huijser *et al.* 2008). Together, identifying spatial and temporal patterns of wildlife crossing activity, based on habitat characteristics and species behavior and life history (e.g., Beaudry, Demaynadier & Hunter 2008; Beaudry, Demaynadier & Hunter 2010), should inform the development of management priorities and cost-effective mitigation efforts.

FOCUS ON MULTIPLE COMPONENTS OF CONSERVATION ISSUES

We focused our work on addressing a human-wildlife problem with complex elements that are representative of most conservation issues today. Terrapins are believed to be declining across the majority of their range due to multiple anthropogenic threats including habitat development, crab pot mortality, subsidized predators, and roads (Roosenburg 1991; Gibbons *et al.* 2001; Grosse *et al.* 2011; Crawford *et al.* 2014a; Isdell *et al.* 2015; Chambers & Maerz *in press;* Maerz, Seigel & Crawford *in press*). Threats to terrapins are often stage- or sex-specific, such as crab pots posing a threat to adult males and juveniles (Roosenburg *et al.* 1997; Chambers

& Maerz *in press*). We used multiple ecological studies (Chapters 2 and 4) to estimate the effects of two road-associated threats to different terrapin life stages on the JIC: road mortality of adult females and predation of nests. In Chapter 4, we showed that management strategies that included complementary actions targeting multiple threats, stage classes, and demographic rates resulted in higher terrapin population growth rates and persistence probabilities. For example, strategies performed well by including roadside barriers to prevent terrapins from crossing at discrete hot spots, flashing warning signage to increase terrapin survival across broader scales, and predator removal and vegetation clearing to influence nest survival and hatchling sex ratios. These results support previous conservation studies showing mitigation of multiple threats was required to ensure the viability of declining populations, and that failure to address challenging threats such as road mortality will undermine the effectiveness of broader strategies (Marschall & Crowder 1996; Rhodes *et al.* 2011; Crawford *et al.* 2014a).

Given substantial complexity and uncertainty involved in modeling human-impacted populations, we incorporated multiple data sources in our analysis to produce more accurate estimates. Integrated models afford the flexibility for leveraging information obtained from multiple, partially-overlapping datasets that reduces the bias and uncertainty in any single dataset (Schaub & Abadi 2011). For most species, sampling all sexes, stages, or other relevant groups necessary to include in population models will require multiple techniques. Specifically, we used mark-recapture under a robust design to sample males and females in tidal creeks and mark-recapture-recover of live and dead females on the JIC using road surveys. Integrated models offer a fruitful approach for estimating population dynamics for rare, cryptic, and declining species since sparse datasets are often all that are available. However, these models have not been extensively used in conservation applications (Schaub & Abadi 2011); thus, our study provides a successful example for using an integrated modeling framework to generate robust predictions for making management decisions even under considerable complexity and uncertainty.

Our work in Chapters 3 and 5 recognized that multiple stakeholders were tied to conservation management of the JIC and formally incorporated their attitudes, values, and perspectives in the decision making process. We measured attitudes of different groups using the JIC in Chapter 3, and we used structured decision making (SDM) to evaluate shared and conflicting social, ecological, and economic objectives of the full suite of stakeholders in Chapter 5. For our study, the SDM process was an effective tool for navigating a complex conservation issue and incorporating multiple perspectives so that final management recommendations could transparently reflect differing values. Including multiple objectives during this process allowed stakeholders to develop a shared understanding of the decision context, transparently acknowledge existing tradeoffs, and produce recommendations that optimally addressed conservation and socioeconomic goals for Jekyll Island. SDM has been applied to a variety of conservation contexts with multiple socioecological objectives (e.g., Conroy *et al.* 2008; Gregory & Long 2009; Converse *et al.* 2013). Our work contributes another example of its effective use for addressing human-wildlife conflicts and is the first application of SDM for road management.

INVOLVE PEOPLE IN CONSERVATION SOLUTIONS

Although there is now ubiquitous agreement that human practices contribute to declines of natural resources, conservation science has been slow to incorporate people, and their behaviors, into management solutions (Cowling 2014). Our work placed equal emphases on ecological and human dimensions of the human-wildlife conflict and potential solutions for roads. Most SDM applications in natural resources have focused primarily on biological (population) objectives. Social objectives (e.g., maximize public satisfaction, agency public relations) are less frequently considered and, if included, are evaluated using coarse metrics and expert opinion (e.g., Converse *et al.* 2013). Our study is one of the first to integrate human dimensions research, capturing the preferences of large stakeholder groups, in an SDM framework. Measuring attitudes and management preferences was especially informative for decisions for the JIC since conflict existed between different groups of stakeholders in several areas; we expect this is the norm, and not the exception, for multi-stakeholder conservation issues. We also found that SDM provided an effective process for working constructively with all parties with a stake in management decisions. Stakeholders, some of whom have never worked together previously, engaged in dialogue and contributed to the decision making process during the all-day workshop. It is our perspective that this collaborative process built social capital between groups on Jekyll Island that facilitated equitable management decisions. By social capital, we mean that groups developed a shared understanding of the problem and each other's perspectives, connectedness that enabled information exchange and leveraging of resources, and feelings of trust that encouraged compromise through engagement in the SDM process (Uphoff 2000; Pretty & Smith 2004). Overall, the SDM process provided a means to prevent unexpected consequences of failing to include stakeholders and consider their views and objectives, and using this collaborative approach with all parties yielded optimal conservation solutions.

Not only did we involve people in the decision making process, our work described promising solutions that achieved conservation outcomes by changing human behavior. In Chapter 4, we found that terrapin survival increased significantly when we implemented flashing warning signage designed to alert drivers during peak periods of crossing activity. By using a modified before-after-control-impact design to test the effects of flashing signage, this study provides strong evidence that the observed increase in terrapin survival was attributable to increased awareness of drivers when signs were flashing that reduced the frequency of terrapin-vehicle collisions. In the context of road management, signage and awareness campaigns designed to change driver behaviors may be more logistically feasible and less costly than building structures to restrict animal access on the road, especially in road networks where wildlife-vehicle collisions are spatially diffuse. We also evaluated the efficacy of behavior change interventions that used social marketing in Chapter 6. Rare's integrated model of behavior change significantly increased community's pro-conservation knowledge, attitudes, and behaviors across diverse contexts such as small-scale fisheries, endangered species poaching, and reciprocal watershed agreements (Jenks, Vaughan & Butler 2010; DeWan *et al.* 2013; Green *et al.* 2013). These results highlight the importance of incorporating behavioral theory into traditional conservation programs and the potential value of applying social marketing to facilitate behavior change and mitigate threats to biodiversity across the globe. Together, the body of work in this dissertation demonstrates effective approaches for integrating people as active participants in decision making and management interventions that promote equitable conservation outcomes.

LIMITATIONS AND FUTURE DIRECTIONS

Diamondback terrapins and road management

We used complex models of the diamondback terrapin population in Chapter 4 that advanced previous terrapin studies but also highlight persistent knowledge gaps for this species. These models realistically represented demographic processes specific to a road context, but they included several parameters that were estimated with substantial uncertainty from empirical data or expert opinion. In Chapter 2, we estimated nest success using simulated nests with chicken eggs since detecting terrapin nests on roadsides is difficult unless a female is observed during the nesting process. Simulated nests provided a reasonable approach for measuring predation pressure given this challenge, but future work should measure true terrapin nest survival across different roadside habitats, and how rates vary annually, with systematic surveying. Uncertainty around demographic estimates for adult terrapins existed in this and other studies due to low rates of detection and availability as individuals use complex habitats and may temporarily move out of the sampling area (Gibbons *et al.* 2001; Tucker, Gibbons & Greene 2001; Mitro 2003; Grosse 2006). We seined for terrapins using mark-recapture methods under a robust design to estimate adult survival and site fidelity, but it may be necessary to employ complementary sampling techniques or longer periods of data collection to account for low detection and temporary movement. Multiple sampling methods have been tested or are in developmental phases for terrapins, including the use of modified crab traps (Roosenburg *et al.* 1997), photographic mark-resighting using drones, or presence-absence surveys using citizen science programs, which could yield informative datasets. Given sampling challenges for the diamondback terrapin and similarly cryptic species of conservation concern, future studies should employ integrated modeling approaches, similar to those used in Chapter 4, to most efficiently use all available data and increase the accuracy and precision of parameter estimates (Schaub & Abadi 2011). Additionally, most terrapin studies have focused on populations under significant anthropogenic threats (e.g., chronic crab pot mortality: Dorcas, Willson & Gibbons 2007), but few have examined population dynamics in the absence of these stressors (Mitro

2003). Obtaining accurate, baseline estimates for non-threatened terrapin populations will be essential in evaluating the impacts of threats in other populations and setting management goals.

Uncertainty around terrapin population estimates was compounded in our population viability models in Chapter 4 as we used expert opinions for the effects of many management actions. Population monitoring programs can be used to replace expert opinions with empirical data and improve decision models for the JIC using an adaptive management framework (Conroy & Peterson 2013). Many actions considered were permanent (e.g., roadside barriers, on-road signage) and may be difficult to modify regardless of future information. Other actions could be applied seasonally (e.g., predator removal, vegetation clearing, awareness programs), allowing models to be updated with new data between years and inform recurrent decisions. Monitoring the impacts of permanent and seasonal actions on population outcomes will provide evidence for promoting effective actions for other at-risk terrapin populations and species.

It is important to consider the ecological and social context of our study site on Jekyll Island when using the results of this dissertation to inform management at other sites where terrapins are declining or for other species. We found that hybrid roadside barriers significantly reduced the number of terrapins emerging on the road when deployed at hot spots of nesting activity. Barriers were fairly costly and labor-intensive to construct and maintain, so identifying discrete, priority areas was essential for our work to be cost-effective. However, previous research could not identify habitat features useful for predicting nesting (road crossing) hot spots (Crawford *et al.* 2014b). Therefore, site-specific monitoring of road mortality is necessary to identify hot spots on other high-traffic coastal roads prior to installation of any barriers. Alternatively, warning signage programmed to flash around high tide may be readily implementable in areas of high terrapin-vehicle collisions. Other studies across the terrapin's range have observed temporal peaks of nesting activity associated with high tide (Burger & Montevecchi 1975; Feinberg & Burke 2003), so the scheduled high tide likely represents a reliable cue to predict hot moments and implement targeted management such as signage. Ultimately, we advocate that researchers and managers grappling with a pervasive human-wildlife issue, such as roads, should follow approaches used here and in other studies (Langen, Ogden & Schwarting 2009; Beaudry, Demaynadier & Hunter 2010; Cureton & Deaton 2012) to identify threat hot spots and hot moments in order to establish management priorities. Furthermore, linking species life history and behavior to these peaks can be used to develop targeted and innovative management strategies (e.g., flashing lights that coincide with daily nesting activity around high tide).

We only considered two road-associated threats (road mortality and nest predation) common to our study site, but crab traps, habitat development, pollution, and other widespread threats continue to contribute to terrapin population declines across their range. Terrapin management initiatives should consider and prioritize all threats occurring within the scale of interest. For state-wide or regional efforts, such as those conducted by state natural resource agencies or special interest groups (e.g., Diamondback Terrapin Working Group), priorities for terrapin management may need to vary at different sites or scales. For example, Grosse *et al.* (2011) found that lower terrapin population densities across the Georgia Coast were more strongly and negatively associated with crabbing pressure than road density. However, the authors cautioned that mortality on high-traffic causeways would likely contribute to localized declines, which was supported in our previous (Crawford *et al.* 2014a) and current research on the JIC. Since multiple threats disproportionately contribute to terrapin declines at different scales, coordinated efforts among researchers and managers will be necessary to predict the effects of threats and management actions and reach optimal decisions – ideally through a formal process such as SDM.

Regarding the social dimensions of our work, most groups, including visitors and residents, valued the protection of local wildlife and were generally accepting to management actions, which is likely due in part to Jekyll Island's status as a state park that attracts those who support conservation efforts. Because state parks, national seashores (e.g., Cumberland Island, GA, Assateague Island, MD), and nature centers are common in coastal areas throughout the terrapin's range, the attitudes of road users in our study may be representative of other sites. Still, measuring local attitudes and management preferences should occur before implementing actions at other sites even if actions were effective and acceptable to JIC users. While we expect flashing signage to be a viable solution at other sites since terrapin nesting behavior is predictable across the species range, driver behavior is less predictable. It is possible the effectiveness of signage on the JIC was influenced by the proportion of drivers that valued wildlife and increased their awareness in response to visual warnings. Signage deployed on roads used by fewer drivers with pro-conservation values may be less effective at changing behaviors. Therefore, deployment of signage and awareness campaigns should be coupled with research to directly measure changes in drivers' attitudes and behaviors at other sites.

Structured decision making

While SDM has yielded efficient and defendable decisions for this and many other conservation contexts, we must acknowledge several important limitations and areas for future growth in the SDM process. SDM is designed to provide a venue and process where all relevant stakeholders are given a "seat at the table," which allows them to contribute perspectives and coproduce satisfactory decisions. This is intended to limit political contention after decisions are implemented (Glicken 2000; Goodwin & Wright 2007). We identified stakeholder groups to be included in the SDM process using informal local knowledge and discussions with Jekyll Island managers. Although we are confident our approach included the full suite of stakeholder groups, performing more rigorous assessments beforehand (i.e., stakeholder analysis: Brugha & Varvasovszky 2000; Glicken 2000) would provide a better approach for identifying all relevant stakeholders and characterizing groups' demographics and interconnectedness. Information about stakeholder dimensions is necessary to ensure an efficient, collaborative process and anticipate areas of contention. Since most SDM practitioners currently come from natural science backgrounds, future applications of SDM should draw on the technical expertise of human dimensions and social scientists to examine the social dimensions of conservation issues before, during, and after the decision making process.

While all stakeholder groups were given a seat at the table in our study, the nature of the SDM process limits the ability of all perspectives within stakeholder groups to be represented. Due to logistical constraints, not all group representatives were involved in each stage of the SDM process. We met with JIA and GSTC staff to produce a provisional problem statement and set of fundamental objectives that were later approved by the full team. We also noticed that representatives from public groups did not contribute significantly to brainstorming of management alternatives, likely due to their lack of technical knowledge about wildlife management. Only CTM, JCM, and I were involved in parameterizing the decision model and conducting the final analysis. Altogether, we acknowledge that all groups did not contribute to all stages, but all groups reviewed and approved the products of each stage before the team advanced to the next SDM stage. We used survey data to incorporate management preferences of large stakeholder groups (e.g., visitors, residents) and supplement the input of representatives from these groups during and after the SDM workshop. Preferences for certain objectives or actions were often variable within a group (see Chapter 3), and we averaged these values when modeling overall satisfaction. Using average preferences for groups is a practical method for evaluating consequences of actions, but we caution that substantial political contention can arise when even a small but vocal subset of individuals differs from the majority regarding a management decision's acceptability. Specifically in the context of Jekyll Island, there has been growing contention between certain resident activist groups and JIA regarding management issues such as deer culling (B. Carswell, pers. comm.), and we were asked not to invite specific individuals to represent these groups at the workshop due to their history of undermining collaborative efforts and unwillingness to compromise. Establishing a set of stakeholder representatives that can engage productively and respectfully is a valid prerequisite for SDM to efficiently resolve conservation issues; however, the exclusion of certain actors, groups, or perspectives signals that SDM is not always a fully inclusive process. The expected benefits and potential roadblocks from excluding these voices is not impractical but needs to be carefully considered by SDM practitioners and decision makers on a context-specific basis.

Similar to other studies (e.g., Converse *et al.* 2013), we applied SDM to a challenging conservation issue where multiple, socioecological objectives must be valued and compared in order to identify optimal decisions. This approach converts objectives on different scales to common units representing their levels of importance to different stakeholders. This practice is difficult to assess in many cases (e.g., groups' values of achieving a 10% reduction in the probability of extinction for a species vs. a $100,000 increase in project costs). Indeed, merging and comparing many distinct values is the subject of strong criticism (Hirsch *et al.* 2011; McShane *et al.* 2011). Multiple, competing objectives may be incommensurable to the point that pooling them into a single metric fails to adequately capture what stakeholders truly value (Hirsch *et al.* 2011; Vercoe *et al.* 2014), which can lead to disgruntled parties even when the traditional steps of SDM are followed. Additionally, decision analytic techniques are used to achieve optimal compromises among groups' values across objectives. In doing so, applications of SDM may identify a final "best" decision that is not the preferred choice for many, or any, groups. We note that this was not the case in our study: the best decision overall was also most preferred by each stakeholder group. However, tradeoffs emerged between objectives: the best decision overall did not perform best, relative to other strategies, on any single objective. Developing a metric with SDM for the degree to which strategies meet a full suite of objectives is a pragmatic and transparent approach for producing optimal recommendations; however, this process requires careful consideration of the nature of these objectives and how they are compared. Practitioners should clearly communicate to stakeholders that SDM does not often provide "win-win" solutions, and instead, acknowledge that underneath any "optimal" conservation decision are costs and benefits that are not distributed uniformly across stakeholders or objectives (ACSC 2011).

Despite certain limitations, SDM provides a transparent process for producing defendable decisions that should be developed in several ways and applied across conservation contexts. First, our study demonstrates a successful example of incorporating human dimensions data into decision models to more accurately estimate social outcomes such as public satisfaction, which have been omitted or coarsely estimated with expert opinion in most other SDM applications. Elicitation of values and attitudes can be conducted easily, even for large, diffuse stakeholder groups, using online or in-person surveys (Vaske 2008), and this process should be informed by consultation with experts in human dimensions and social sciences. Second, roads represent a pervasive and wicked conservation issue that requires strategic planning with multiple stakeholders. Our study describes the first application of SDM for road management and provides a model for addressing other contexts where roads impact species of conservation concern. The majority of roads exist on public lands and are heavily traveled by people representing a variety of demographics and values. Therefore, the steps of SDM employed in this study are likely applicable to other road contexts that require consideration of multiple stakeholders, including public groups, and objectives. We expect SDM applications for most road issues will consider wildlife conservation, human safety, public satisfaction, and cost objectives, similar to our study.

Lastly, SDM (described in Chapter 5) and behavior change (described in Chapter 6) represent two promising but seldom-overlapping spheres of research and practice for addressing conservation issues. We advocate that future interdisciplinary efforts should integrate the theories, tools, and methods of behavior change into SDM. The scope of potential actions evaluated through SDM has been mostly limited to two types of strategies: 1) direct management of natural resources, such as improvement of core habitats (Gregory & Long 2009) or reintroduction of endangered species (Converse *et al.* 2013), and 2) top-down formation of regulatory policy (e.g., legislature preventing species harvest: Nichols & Williams 2006). Meanwhile, conservation behavior change focuses exclusively on a third type of strategy: bottom-up, voluntary interventions designed to replace destructive behaviors with sustainable ones. There have been increasing calls to incorporate human behavior into conservation solutions (Schultz 2011; St John, Edwards-Jones & Jones 2011; Cowling 2014), and we showed that behavior change campaigns are effective for producing pro-conservation outcomes. Thus, leveraging the strengths of SDM and behavior change should yield an innovative approach that will allow for meaningful comparisons across three types of strategies for resolving conservation issues. Integration, instead of isolation, of SDM and behavior change should advance our ability to identify optimal solutions, make better investments of limited resources, and address complex conservation challenges in a world where behavior-driven threats continue to impact nature.

REFERENCES

ACSC (2011) *Advancing conservation in a social context: Working in a world of trade-offs,* Final Report edn. Available from Available online at: http://cicr.uga.edu/acsc/

Aresco, M.J. (2005) The effect of sex-specific terrestrial movements and roads on the sex ratio of freshwater turtles. *Biological Conservation,* **123,** 37-44.

Beaudry, F., Demaynadier, P.G. & Hunter, M.L. (2008) Identifying road mortality threat at multiple spatial scales for semi-aquatic turtles. *Biological Conservation,* **141,** 2550-2563.

Beaudry, F., Demaynadier, P.G. & Hunter, M.L. (2010) Identifying hot moments in road-mortality risk for freshwater turtles. *Journal of Wildlife Management,* **74,** 152-159.

Brugha, R. & Varvasovszky, Z. (2000) Stakeholder analysis: a review. *Health policy and planning,* **15,** 239-246.

Burger, J. & Montevecchi, W.A. (1975) Nest Site Selection in Terrapin *Malaclemys terrapin*. *Copeia,* **1975,** 113-119.

Butler, J., Heinrich, G. & Seigel, R. (2006) Third workshop on the ecology, status, and conservation of diamondback terrapins (*Malaclemys terrapin*): results and recommendations. *Chelonian Conservation and Biology,* **5,** 331-334.

Caswell, H. (1989) *Matrix population models: construction, analysis, and interpretation*. Sinauer Associates, Sunderland, U.S.A.

Chambers, R.M. & Maerz, J.C. (in press) Terrapin bycatch in the blue crab fishery. Ecology and conservation of the diamondback terrapin (eds W.M. Roosenburg & V.S. Kennedy). Johns Hopkins University Press, Baltimore, MD.

Conroy, M.J., Barker, R.J., Dillingham, P.W., Fletcher, D., Gormley, A.M. & Westbrooke, I.M. (2008) Application of decision theory to conservation management: recovery of Hector's dolphin. *Wildlife Research,* **35,** 93-102.

Conroy, M.J. & Peterson, J.T. (2013) *Decision making in natural resource management: a structured, adaptive approach*. Wiley,

Converse, S.J., Moore, C.T., Folk, M.J. & Runge, M.C. (2013) A matter of tradeoffs: reintroduction as a multiple objective decision. *The Journal of Wildlife Management,* **77,** 1145-1156.

Cowling, R.M. (2014) Let's Get Serious About Human Behavior and Conservation. *Conservation Letters,* **7,** 147-148.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A. & Norton, T.M. (2014a) Estimating the consequences of multiple threats and management strategies for semi-aquatic turtles. *Journal of Applied Ecology,* **51,** 359-366.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A., Norton, T.M. & Albeke, S.E. (2014b) Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology,* **51,** 367-375.

Cureton, J.C. & Deaton, R. (2012) Hot moments and hot spots: identifying factors explaining temporal and spatial variation in turtle road mortality. *Journal of Wildlife Management,* **76,** 1047-1052.

DeWan, A., Green, K., Li, X. & Hayden, D. (2013) Using social marketing tools to increase fuel-efficient stove adoption for conservation of the golden snub-nosed monkey, Gansu Province, China. *Conservation Evidence,* **10,** 32-36.

Dorcas, M.E., Willson, J.D. & Gibbons, J.W. (2007) Crab trapping causes population decline and demographic changes in diamondback terrapins over two decades. *Biological Conservation,* **137,** 334-340.

Feinberg, J.A. & Burke, R.L. (2003) Nesting ecology and predation of diamondback terrapins, *Malaclemys terrapin,* at Gateway National Recreation Area, New York. *Journal of Herpetology,* **37,** 517-526.

Gibbons, J.W., Lovich, J.E., Tucker, A.D., Fitzsimmons, N.N. & Greene, J.L. (2001) Demographic and ecological factors affecting conservation and management of the Diamondback Terrapin (Malaclemys terrapin) in South Carolina. Chelonian Conservation and Biology, **4,** 66-74.

Gilliand, S.C., Chambers, R.M. & LaMar, M.D. (2014) Modeling the effects of crab potting and road traffic on a population of diamondback terrapins. *Proceedings of the Symposium on BEER.*

Glicken, J. (2000) Getting stakeholder participation 'right': a discussion of participatory processes and possible pitfalls. *Environmental Science & Policy,* **3,** 305-310.

Goodwin, P. & Wright, G. (2007) *Decision analysis for management judgment*. John Wiley & Sons, Chichester, United Kingdom.

Green, K.M., DeWan, A., Arias, A.B. & Hayden, D. (2013) Driving adoption of payments for ecosystem services through social marketing, Veracruz, Mexico. *Conservation Evidence,* **10,** 48-52.

Gregory, R. & Long, G. (2009) Using structured decision making to help implement a precautionary approach to endangered species management. *Risk Analysis,* **29,** 518-532.

Grosse, A.M. (2006) *Assessment of the effects of roads and crabbing pressures on diamondback terrapin populations*. M.S. thesis, University of Georgia.

Grosse, A.M., Maerz, J.C., Hepinstall-Cymerman, J. & Dorcas, M.E. (2011) Effects of roads and crabbing pressures on diamondback terrapin populations in coastal Georgia. *Journal of Wildlife Management*, **75,** 762-770.

Hart, K.M. (2005) *Population biology of diamondback terrapins (Malaclemys terrapin): defining and reducing threats across their geographic range*. PhD thesis, Duke University.

Hirsch, P.D., Adams, W.M., Brosius, J.P., Zia, A., Bariola, N. & Dammert, J.L. (2011) Acknowledging Conservation Trade-Offs and Embracing Complexity. *Conservation Biology,* **25,** 259-264.

Huijser, M.P., McGowen, P.T., Fuller, J., Hardy, A. & Kociolek, A. (2008) *Wildlife-vehicle collision reduction study: Report to Congress*. Federal Highway Administration, Washington, DC.

Isdell, R.E., Chambers, R.M., Bilkovic, D.M. & Leu, M. (2015) Effects of terrestrial–aquatic connectivity on an estuarine turtle. Diversity and Distributions, **21,** 643-653.

Jenks, B., Vaughan, P.W. & Butler, P.J. (2010) The evolution of Rare Pride: Using evaluation to drive adaptive management in a biodiversity conservation organization. *Evaluation and Program Planning,* **33,** 186-190.

Langen, T.A., Machniak, A., Crowe, E.K., Mangan, C., Marker, D.F., Liddle, N. & Roden, B. (2007) Methodologies for surveying herpetofauna mortality on rural highways. *Journal of Wildlife Management,* **71,** 1361-1368.

Langen, T.A., Ogden, K.M. & Schwarting, L.L. (2009) Predicting hot spots of herpetofauna road mortality along highway networks. *Journal of Wildlife Management,* **73,** 104-114.

Maerz, J.C., Seigel, R.A. & Crawford, B.A. (in press) Terrapin conservation: Mitigating habitat loss, road mortality, and subsidized predators. Ecology and conservation of the diamondback terrapin (eds W.M. Roosenburg & V.S. Kennedy). Johns Hopkins University Press, Baltimore, MD.

Marschall, E.A. & Crowder, L.B. (1996) Assessing population responses to multiple anthropogenic effects: a case study with brook trout. *Ecological Applications,* **6,** 152-167.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L. & Pulgar-Vidal, M. (2011) Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation,* **144,** 966-972.

Mitro, M.G. (2003) Demography and viability analyses of a diamondback terrapin population. *Canadian Journal of Zoology-Revue Canadienne De Zoologie,* **81,** 716-726.

Mumme, R.L., Schoech, S.J., Woolfenden, G.E. & Fitzpatrick, J.W. (2000) Life and death in the fast lane: demographic consequences of road mortality in the Florida Scrub-Jay. *Conservation Biology,* **14,** 501-512.

Munscher, E.C., Kuhns, E.H., Cox, C.A. & Butler, J.A. (2012) Decreased nest mortality for the Carolina diamondback terrapin (*Malaclemys terrapin centrata*) following removal of raccoons (*Procyon lotor*) from a nesting beach in northeastern Florida. *Herpetological Conservation and Biology,* **7,** 176-184.

Nichols, J.D. & Williams, B.K. (2006) Monitoring for conservation. *Trends in Ecology & Evolution,* **21,** 668-673.

Pretty, J. & Smith, D. (2004) Social capital in biodiversity conservation and management. *Conservation Biology,* **18,** 631-638.

Putman, R.J. (1997) Deer and road traffic accidents: Options for management. *Journal of Environmental Management*, **51,** 43-57.

Rhodes, J.R., Ng, C.F., de Villiers, D.L., Preece, H.J., McAlpine, C.A. & Possingham, H.P. (2011) Using integrated population modelling to quantify the implications of multiple threatening processes for a rapidly declining population. *Biological Conservation,* **144,** 1081-1088.

Roosenburg, W.M. (1991) The diamondback terrapin: Population dynamics, habitat requirements, and opportunities for conservation. New Perspectives in the Chesapeake System: A Research and Management Partnership. Proceedings of a Conference. CRC Publ, pp. 227-234.

Roosenburg, W.M., Cresko, W., Modesitte, M. & Robbins, M.B. (1997) Diamondback terrapin (*Malaclemys terrapin*) mortality in crab pots. *Conservation Biology,* **11,** 1166-1172.

Roosenburg, W.M. & Place, A.R. (1994) Nest predation and hatchling sex ratio in the diamondback terrapin: implications for management and conservation. *Tortoise Sustainable Coastal Watershed: The Chesapeake Experiment. Chesapeake Research Consortium, Publication***,** 65-70.

Row, J.R., Blouin-Demers, G. & Weatherhead, P.J. (2007) Demographic effects of road mortality in black ratsnakes (*Elaphe obsoleta*). *Biological Conservation,* **137,** 117-124.

Schaub, M. & Abadi, F. (2011) Integrated population models: a novel analysis framework for deeper insights into population dynamics. *Journal of Ornithology,* **152,** 227-237.

Schultz, P. (2011) Conservation means behavior. *Conservation Biology,* **25,** 1080-1083.

St John, F.A., Edwards-Jones, G. & Jones, J.P. (2011) Conservation and human behaviour: lessons from social psychology. *Wildlife Research,* **37,** 658-667.

Tucker, A.D., Gibbons, J.W. & Greene, J.L. (2001) Estimates of adult survival and migration for diamondback terrapins: conservation insight from local extirpation within a metapopulation. *Canadian Journal of Zoology,* **79,** 2199-2209.

Uphoff, N. (2000) Understanding social capital: learning from the analysis and experience of participation. *Social capital: A multifaceted perspective* (eds I. Serageldin & P. Dasgupta), pp. 215-249. Oxford University Press, Oxford, UK.

Vaske, J.J. (2008) *Survey research and analysis: Applications in parks, recreation and human dimensions*. Venture Publishing, Inc., State College, PA.

Vercoe, R.A., Welch-Devine, M., Hardy, D., Demoss, J.A., Bonney, S.N., Allen, K., Brosius, P., Charles, D., Crawford, B., Heisel, S., Heynen, N., de Jesús-Crespo, R.G., Nibbelink, N., Parker, L., Pringle, C., Shaw, A. & Van Sant, L. (2014) Acknowledging Trade-offs and Understanding Complexity: Exurbanization Issues in Macon County, North Carolina. *Ecology and Society,* **19,** 23.

APPENDIX A

SURVEY INSTRUMENT FOR MEASURING ATTITUDES AND VALUES OF JEKYLL ISLAND PATRON GROUPS

Section A: Past experience with Jekyll Island

A1. What best describes your connection with Jekyll Island? (*Check ONE answer*)

☐ Visitor
☐ Resident (***Skip to A3***)
☐ Island Employee (***Skip to A3***)
☐ Resident AND Employee (***Skip to A3***)

A2. Including this trip, how many times have you recently visited Jekyll Island?

________________# trips over last 12 months

__________________# trips over last 3 years

A3. Which of the following activities do you typically do on Jekyll Island? (*Check ALL that apply*)

☐ Beach activities
☐ Fishing
☐ Historical sightseeing
☐ Wildlife viewing
☐ Golfing
☐ Biking/hiking
☐ Camping
☐ Other_________________
☐ Georgia Sea Turtle Center
☐ Tidelands Nature Center
☐ Summer Waves Waterpark
☐ Other______________________

A4. Please answer the following questions. (*Check ONE box per ROW*)

	No	**Unsure**	**Yes**
Could you identify a terrapin?	☐	☐	☐
Have you witnessed a terrapin hit on the Causeway?	☐	☐	☐
Have you hit a terrapin on the Causeway?	☐	☐	☐

A5. Indicate your level of agreement with the following statements. (*Circle ONE number per ROW*)

	Strongly Disagree ⟵ **1**	**2**	**3**	**4**	⟶ **Strongly Agree** **5**
The number of terrapins living near the Causeway has declined in recent years.	1	2	3	4	5
We should protect terrapins from being struck by vehicles on the Causeway.	1	2	3	4	5
We should protect terrapin nests from raccoon predators.	1	2	3	4	5
Protecting terrapins should NOT be a priority when managing the Causeway.	1	2	3	4	5

Section B: Causeway management and wildlife

Of many potential management options, you may not be familiar with the following:
Creating roadside barriers and/or nest boxes that would intercept adult turtles before they cross the road. These may detract from scenic marsh views. Clearing roadside hedges would increase marsh views and give terrapins more nesting habitat.

B1. How acceptable is it for the following management actions to occur on the Causeway? (*Please circle ONE number per ROW*)

	Not at All Acceptable ⟷ **Highly Acceptable**				
	1	**2**	**3**	**4**	**5**
Reduce speed limit	1	2	3	4	5
Install speed bumps	1	2	3	4	5
Install flashing yellow lights to warn drivers of terrapins on the road	1	2	3	4	5
Place short fences (1-ft in height) along roadsides	1	2	3	4	5
Place concrete barriers along roadsides	1	2	3	4	5
Construct more nest boxes in plain sight	1	2	3	4	5
Construct more hidden nest boxes	1	2	3	4	5
Clear roadside hedges	1	2	3	4	5
Relocate raccoons	1	2	3	4	5
Lethally (humanely) remove raccoons	1	2	3	4	5
Do nothing	1	2	3	4	5

B2. Please indicate your level of agreement with the following statements. (C*ircle ONE number per ROW*)

	Strongly Disagree ⟷ **Strongly Agree**				
	1	**2**	**3**	**4**	**5**
Terrapins are a valuable part of nature.	1	2	3	4	5
We should maintain terrapin survival.	1	2	3	4	5
We should maintain views of the marshland from the Causeway regardless of impacts on terrapins.	1	2	3	4	5
We should maintain the current speed limit regardless of the impact on terrapins.	1	2	3	4	5
Terrapins are an important component to Jekyll Island's scenic beauty.	1	2	3	4	5
Protecting terrapins is important to preserve the natural integrity of Jekyll Island.	1	2	3	4	5
I think it is inhumane to kill terrapins on the Causeway.	1	2	3	4	5
Terrapins have as much right as people to exist.	1	2	3	4	5
Management should focus on doing what is best for island patrons over what is best for terrapins.	1	2	3	4	5

B3. How concerned would you be for the following if you were in a wildlife-vehicle collision? (*Please circle ONE number per ROW*)

	Not at All Concerned ⟵			⟶	**Extremely Concerned**
	1	**2**	**3**	**4**	**5**
Impact to human safety	1	2	3	4	5
Damage to vehicle	1	2	3	4	5
Impact to wildlife	1	2	3	4	5

B4. How upset would you be if you struck each of the following animals while driving? (*Please circle ONE number per ROW*)

	Not at All Upset ⟵			⟶	**Extremely Upset**
	1	**2**	**3**	**4**	**5**
Bear	1	2	3	4	5
Deer	1	2	3	4	5
Raccoon	1	2	3	4	5
Snake	1	2	3	4	5
Turtle	1	2	3	4	5
Bird	1	2	3	4	5
Squirrel	1	2	3	4	5
Dog	1	2	3	4	5
Cat	1	2	3	4	5

Section C: Demographics

These questions will help us to ensure that the people we are surveying are representative of all people that travel on the Jekyll Island Causeway. **All answers will be kept strictly confidential.**

C1. What is your age? ________________years

C2. What is your gender? ___________Male ____________Female

C3. Which of the following category best describes your race/ethnicity? (*Check ALL that apply*)

☐ Caucasian

☐ African American

☐ Hispanic, Latino, or Spanish

☐ Asian or Pacific Islander

☐ American Indian

☐ Other

C4. What is your highest level of education? (*Check ONE answer*)

☐ High school not completed

☐ High school completed

☐ Some college or technical school

☐ College degree or higher

C5. What is your zip code? _______________

Please use the space provided below for any additional comments.

Thank you for completing this survey. If you have any additional questions, please contact:

Brian Crawford – (301) 529-8206; bcrawfor@uga.edu

Additional questions or problems regarding your rights as a study participant should be addressed to The Chairperson, Institutional Review Board, University of Georgia, 629 Boyd Graduate Studies Research Center, Athens, Georgia 30602-7411;
Telephone (706) 542-3199; E-Mail Address irb@uga.edu

APPENDIX B

DETAILED DESCRIPTION OF DEMOGRAPHIC MODEL AND POPULATION PROJECTION MODEL IN CHAPTER 4

PARAMETER ESTIMATION

Robust Design submodel

For the Robust Design submodel, we created a set of candidate models with a range of alternative structures and compared them using Deviance Information Criterion (DIC) (Spiegelhalter *et al.* 2002). The model structures most supported by the data were then used in the final integrated model. We specified the same structure for certain parameters among all models within candidate sets in order to allow for estimation of parameters needed for the population viability analysis (PVA) and enable joint parameter estimation with the female-only multistate model. We modeled the logit of survival (φ) for individuals in the Robust Design dataset as a function of sex (g), year (y), and other covariate effects using the relationship

$$\text{logit}(\varphi_{g,y}) = \mu_g + \varepsilon_{g,y} + \beta_{per}X_y \qquad (1)$$

where μ_g (sex-specific intercept), $\varepsilon_{g,y}$ (random year effect for each group), and β_{per} (coefficient for the effect of management on survival) are parameters to be estimated from data and X_y is a binary indicator variable set to 0 in years prior to management implementation and 1 after. We modeled entry probabilities across years in the study using a Dirichlet prior where annual entry probabilities (b_y) were drawn from an uninformative gamma distribution $\eta_y \sim Gamma(1,1)$ and then set as $\beta_y = \eta_y / \sum_{j=1}^{Y} \eta_j$. We then re-expressed rates as conditional entry probabilities (b_y), the probability an individual enters the population in year y given that it has not yet entered, as follows:

$$b_1 = \beta_1$$

$$b_y = \frac{\beta_y}{1-\sum_{i=1}^{Y-1}\beta_i}, y = 2, \dots, Y. \qquad (2)$$

We followed Kéry and Schaub (2012) to construct a state-space formulation of the Robust Design submodel where estimation of parameters governing the demographic and observation processes was given by Eqs. 3 and 4, respectively, and were consistent across candidate models:

$$\mathrm{z}_{i,1} \sim Bernoulli(b_1)$$

$$\mathrm{z}_{i,y+1} \mid \mathrm{z}_{i,y} \sim Bernoulli\left(\varphi_{i,y-1} z_{i,y-1} + b_y \prod_{y=1}^{Y-1}(1 - z_{i,y})\right), y = 2, \dots, Y \qquad (3)$$

where the term $\prod_{y=1}^{Y-1}(1 - z_{i,y})$ indicates availability to enter the population, given the individual has not previously entered before time *y*, and

$$w_i \sim Bernoulli(\psi_{RD})$$

$$\mathrm{y}_{i,t,y} \mid \mathrm{z}_{i,y} \sim Bernoulli\left(w_i z_{i,y} p_{i,t}\right). \qquad (4)$$

where the inclusion of parameter w_i in the observation model accounts for zero-inflation of the augmented superpopulation dataset.

We constructed a candidate set of Robust Design models and performed model selection in three stages. In the first stage, we evaluated the relationship between capture (*p*) and recapture (*c*) by creating and comparing a model where these parameters were allowed to vary independently by year and a model where p=c and varied by year. Because the latter model outperformed the former (ΔDIC = 40.9) and recaptures were sparse in our dataset, we proceeded with setting capture equal to recapture in all further models. In the second stage, we varied the parameter structure of capture while keeping all other parameter structures fixed. We specified sex-specific capture probabilities as constant, fixed effect for a daily tide amplitude covariate, random effects for sampling occasion, or the combination of fixed covariate effects and random day effects. We expected tide amplitude to negatively influence capture probability since higher tides create deeper and wider creeks and thus more area that is missed by our fixed-length seining nets. In the third stage, we constructed a candidate set where all models included the best-performing structure for capture but varied in parameter structure of site fidelity (γ). We specified sex-specific site fidelity probabilities as constant, fixed effects for year, or random effects for year.

Multistate Jolly-Seber submodel

We included additional parameters when modeling survival for crossing and non-crossing individuals in the multistate mark-recapture dataset. We modeled the logit of survival (φ) for non-crossing and crossing females as a function of sex (g), year (y), and other covariate effects using the relationships

$$\mathrm{logit}(\varphi_y) = \begin{cases} \mu + \varepsilon_y + \beta_{per}X_y + \beta_{cross} + \beta_{int.nc}X_y & \text{for non} - \text{crossing females} \\ \mu + \varepsilon_y + \beta_{per}X_y - \beta_{cross} + \beta_{int.c}X_y & \text{for crossing females} \end{cases} \tag{5}$$

where μ (female intercept), ε_y (random year effect for females), β_{per} (coefficient for the effect of management on survival), β_{crossr} (coefficient for the effect of crossing status), and β_{int} (coefficient for the interaction effect of management period and crossing status for each group) are parameters to be estimated from data and X_y is a binary indicator variable set to 0 in years prior to management implementation and 1 after.

We followed Kéry and Schaub (2012) to construct the multistate Jolly-Seber submodel where estimation of parameters governing the demographic and observation processes was given by Eqs. 6 and 7, respectively:

$$\mathrm{z}_{i,y+1} \mid \mathrm{z}_{i,y} \sim Multinomial(1, \mathrm{z}_{i,y}\boldsymbol{\Omega}_{\boldsymbol{i,y}}) \qquad (6)$$

where

$$\boldsymbol{\Omega}_{\boldsymbol{i,y}} = \begin{array}{c|cccc} & NYE & Ac & Anc & Ao \\ NYE & 1-b & b(\gamma)(c)(\varphi.c) & b(\gamma)(1-c)(\varphi.nc) & b(1-\gamma)(\varphi.nc) \\ Ac & 0 & \gamma(c)(\varphi.c) & \gamma(1-c)(\varphi.nc) & (1-\gamma)\varphi.nc \\ Anc & 0 & \gamma(c)(\varphi.c) & \gamma(1-c)(\varphi.nc) & (1-\gamma)\varphi.nc \\ Ao & 0 & \gamma(c(\varphi.c) & \gamma(1(\varphi.nc) & (1-\gamma)\varphi.nc \\ Dc & 0 & 0 & 0 & 0 \\ Do & 0 & 0 & 0 & 0 \end{array}$$

$$\begin{array}{cc|} Dc & Do \\ b(\gamma)(1-\varphi.c) & b(1-\gamma)(1-\varphi.nc) + b(\gamma)(1-\varphi.nc) \\ \gamma(\psi_c)(1-\varphi.c) & (1-\gamma)(1-\varphi.nc) + \gamma(1-\psi_c)(1-\varphi.nc) \\ \gamma(1-\psi_c)(1-\varphi.c) & (1-\gamma)(1-\varphi.nc) + \gamma(\psi_c)(1-\varphi.nc) \\ \gamma(1-\varphi.c) & (1-\gamma)(1-\varphi.nc) + \gamma(1-\varphi.nc) \\ 0 & 1 \\ 0 & 1 \end{array}$$

and

$$w_i \sim Bernoulli(\psi_{MS})$$

$$\mathrm{y}_{i,y} \mid \mathrm{z}_{i,y} \sim Multinomial(1, (\mathrm{z}_{i,y}w_i + 4(1-w_i))\boldsymbol{\Theta}_{\boldsymbol{i,y}}) \qquad (7)$$

where the inclusion of parameter *w_i* in the observation model accounts for zero-inflation of the augmented superpopulation dataset and assigns a state of NS, which was specified as state “4” in the dataset, to any non-real member of the superpopulation, and

$$\boldsymbol{\Theta}_{\boldsymbol{i,y}} = \begin{array}{c|cccc} & \multicolumn{4}{c}{\text{Observerd state}} \\ \text{True state} & AC & ANC & DC & NS \\ NYE & 0 & 0 & 0 & 1 \\ AC & p.c & 0 & 0 & 1-p.c \\ ANC & 0 & p.nc & 0 & 1-p.nc \\ AO & 0 & 0 & 0 & 1 \\ DC & 0 & 0 & r & 1-r \\ DO & 0 & 0 & 0 & 1 \end{array}$$

PVA MODEL

We used empirical data or expert opinion to estimate demographic effects of management actions included in each of 20 strategies created through a structured decision making workshop and incorporated each strategy's effects in turn through iterations of the PVA.

We calculated the overall proportion of nesting turtles attempting to cross that were prevented from entering the road (p_{block}) due to management actions (barriers and vegetation removal) included in each strategy as:

$$p_{block} = \sum_{i=1}^{n} p_{turtle_i} * p_{cross} * p_{effect_i}$$

where p_{turtle} is the proportion of all crossing turtles that are in the managed area and encounter management action *i*, p_{cross} is the proportion of nesting females attempting to cross the road, and p_{effect} is the proportion of turtles encountering a device or cleared section of the road that are effectively prevented from crossing. Terrapins on the JIC cross at a disproportionately higher rate in certain areas ("hot spots and warm spots"), and roadside barriers listed in our management strategies were specified at these localized scales or causeway-wide. We used GPS data obtained from 2009-2014 to estimate the proportion of turtles expected to cross at the proposed managed area. We multiplied the proportion of turtles encountering each device by the proportion expected to be prevented from entering the road, which was obtained from direct estimates (see *Estimation of barrier effects*) or expert elicitation. Strategies often included multiple types of barriers at different scales, so we summed individual effects within a given strategy.

We considered vegetation management actions in the forms of clear cutting of roadside hedges in isolation and in conjunction with creation of elevated nesting mounds with or without protective boxes at hot spots of terrapin crossing and nesting activity. Therefore, we calculated the proportion of nesting females ($p_{nesters}$) expected to use habitat type *i* as:

$$p_{nesters_i} = \left(p_{turtle} * p_{mgmt_i}\right) + (1 - p_{turtle}) * p_{nomgmt_i}$$

where p_{turtle} is again the proportion of all crossing turtles that are in the managed area, p_{mgmt} is the proportion of managed area composed of habitat *i*, and p_{nomgmt} is the proportion of unmanaged area (i.e., the remaining portion of the JIC) composed of habitat *i*. We assumed that terrapins nest in each habitat type proportionally to its availability and did not include adult preference for certain habitat types in our estimates. Only a few studies have examined nesting habitat selection in terrapins, which have found weak preferences for open habitat but have observed nesting in a variety of shrubland and anthropogenic habitats (Feinberg and Burke 2003, Roosenberg 1991).

We altered baseline habitat-specific nest success rates obtained from empirical field studies (Grosse et al. 2015) by the expected effects of predator management, vegetation management, and barriers included in strategies based on expert elicitation. We had experts estimate mean nest success given deployment of management actions in isolation or combinations included in each strategy. We altered baseline nest success rates proportionally to the mean effect of management actions. We also used habitat-specific hatchling sex ratios (the proportion of female hatchlings) from empirical field studies (Grosse et al. 2015) in the final population model. While management actions did not directly affect habitat-specific rates, they influenced the average sex ratio by changing the proportion of clutches laid in each habitat type.

Estimation of barrier effects

We estimated effects of nest box barriers (hybrid barriers) on terrapin emergence on the JIC using a modified before-after-control-impact (BACI: Green 1979; Stewart-Oaten, Murdoch & Parker 1986; Skalski & Robson 1992) design over six consecutive years (2009 to 2014). We constructed a hybrid barrier on one roadside at one hot spot of terrapin activity (“Experimental”) while leaving the other two hot spots unaltered (“Control 1” and “Control 2”) prior to the nesting season in 2011, which created two seasons pre- and four post-intervention to analyze. We assumed independence among these sites since all pairwise distances between hot spots were > 1 km and terrapins exhibit nest site fidelity where most individuals attempt to cross the road to nest within 50 – 100 m from previous nesting locations (Szerlag-Egger & McRobert 2007; Crawford *et al.* 2014). The barrier was composed of short fencing and artificial nest boxes designed to prevent females from entering roads and protect nests laid inside boxes from depredation (Buhlmann & Osborn 2011). Barriers were only placed on the south roadside due to logistic constraints and early observations that most terrapins emerged from the southern marsh at that section of the road.

To estimate the effect of barriers on preventing turtles from entering the road, we recorded the total number of terrapins observed on or attempting to cross the road that were located within each site each year. We only included turtles crossing from the south (barricaded) side of the road within the experimental site each year. We analyzed the number of terrapins emerging on the road per year at a site by fitting the following mixed model (hereafter, barrier model) with a Poisson distribution, determined by the same mean and variance (λ):

$$C_{i,j} \sim \text{Poisson}(\lambda_{i,j})$$

$$\log(\lambda_{i,j}) = year_i + site_j + \beta_j X_{i,j} + \varepsilon_{i,j}$$

where $C_{i,j}$ represents the count of terrapins on or attempting to cross the road in year i at site j; $year_i$ is the random year effect; site_j is the random site effect; β_j is the vector of site-specific period (before-after) fixed effects ($X_{i,j} = 1$ in years after the barrier was installed, 0 before intervention); and $\varepsilon_{i,j}$ is the residual error term. Note that by specifying the model this way, estimated β effects represent the pre-post effect size in number of terrapins on the road while accounting for other sources of variation. We calculated e^{β_j} using posterior estimates to represent the mean percent change in terrapins on the road at each control or experimental site relative to pre-intervention periods.

We included year as a random effect, assumed to arise from a zero-centered normal distribution with variance σ_Y^2 , in our model since the annual number of females observed nesting in varied considerably throughout our 6-yr study in a non-systematic pattern. We included the additional random effect for site, assumed to arise from a normal distribution centered on the site mean μ_S with variance σ_S^2 , since sites varied in length (Control 1 = 331 m; Control 2 = 310 m; Experimental = 162 m) and other roadside characteristics (e.g., vegetation composition) that could influence the number of terrapins observed using these areas each year to nest. Terrapins exhibit nest site fidelity, so relative abundance of nesting females from site-to-site is likely to be consistent over time as females return to the same locations. Lastly, we included a residual random effect, assumed to arise from a zero-centered normal distribution with variance σ_E^2 , to improve model fit and account for other unmeasured sources of variation.

We implemented Bayesian mixed modeling using Markov chain Monte Carlo (MCMC) methods to estimate management impacts and fitted the barrier model in WinBUGS 1.4.3 (Spiegelhalter *et al.* 2003) called from R (R Core Team 2013) via the R2WinBUGS package (Sturtz, Ligges & Gelman 2005). We assigned uninformative prior distributions for all fixed effect parameters and hyperparameters governing random effects to represent lack of previous knowledge about management effects. We estimated posterior distributions using 200000 iterations of three chains after discarding the first 50000. We retained every 50th iteration to reduce autocorrelation among samples, which resulted in a total sample size of 9000 from posterior distributions. We assessed convergence for both models by: 1) visually inspecting chain mixing in MCMC plots as well as estimated posterior distributions for evidence of unimodality, and 2) calculating the Brooks-Gelman-Rubin statistic (Brooks & Gelman 1998), which compares within- and between-chain variance. We assessed goodness of fit for the barrier model with a Bayesian P-value (Kéry 2010) – a statistic that compares the discrepancy between observed and simulated data predicted from the model – where model fit is interpreted as best with values near 0.5 and worst as values approach 0 or 1. We based inferences of management effect sizes and direction on posterior means and 95% Bayesian Credible Intervals (BCIs; 2.5th – 97.5th percentile of the distribution). We interpreted parameters as statistically significant when BCIs did not overlap 0.

RESULTS

Parameter estimation

Robust Design submodels varied in fit to the data, measured by DIC, within candidate sets that included different effects for estimating p and γ (Table B1). This allowed us to select the parameter effects for top models and include them in the final integrated model. We present mean (95% BCI) posterior estimates for parameters in the integrated model in Table B2.

Management effects

From 2009 to 2014, we observed 498 terrapins crossing within hot spots that were included in the barrier model. Bayesian mixed models for barrier effects on terrapins showed adequate convergence based on MCMC mixing of chains and Brooks-Gelman-Rubin statistics <1.1 for all parameters. The Bayesian P-value for the barrier model was 0.428 indicating it adequately fit the data.

Posterior mean estimates and 95% Bayesian credible intervals from the barrier model revealed a significant and negative period (before-after) effect for the experimental site (mean β: -1.32, 95% BCI: -2.54 to -0.10) and nonsignificant effects for the two control sites. When comparing the percentage change in terrapin emergence from pre- and post-intervention periods, the model predicted a 61.4% reduction in terrapins on the road at the experimental site after the barrier was installed while terrapin emergence did not change significantly for either control site. We used this estimated management effect as the proportion of turtles prevented from entering the road at sites with hybrid barriers.

PVA model

We present effects of actions contained in each strategy on demographic rates included in the PVA model in Table B3.

REFERENCES

Brooks, S.P. & Gelman, A. (1998) General methods for monitoring convergence of iterative simulations. *Journal of computational and graphical statistics,* **7,** 434-455.

Buhlmann, K.A. & Osborn, C.P. (2011) Use of an artificial nesting mound by wood turtles (*Glyptemys insculpta*): a tool for turtle conservation. *Northeastern Naturalist,* **18,** 315-334.

Crawford, B.A., Maerz, J.C., Nibbelink, N.P., Buhlmann, K.A., Norton, T.M. & Albeke, S.E. (2014) Hot spots and hot moments of diamondback terrapin road-crossing activity. *Journal of Applied Ecology,* **51,** 367-375.

Green, R.H. (1979) *Sampling design and statistical methods for environmental biologists*. Wiley Interscience, Chichester, UK.

Kéry, M. (2010) *Introduction to WinBUGS for ecologists: Bayesian approach to regression, ANOVA, mixed models and related analyses*. Academic Press, Burlington, MA.

Kéry, M. & Schaub, M. (2012) *Bayesian population analysis using WinBUGS: A hierarchical perspective*. Academic Press, Oxford, UK.

R Core Team (2013) R: A language and environment for statistical computing. *R Foundation for Statistical Computing*. Vienna, Austria. Available from http://www.R-project.org/

Skalski, J.R. & Robson, D.S. (1992) *Techniques in Wildlife Investigations: Design and Analysis of Capture Data*. Academic Press, New York.

Spiegelhalter, D., Thomas, A., Best, N. & Lunn, D. (2003) WinBUGS Version 1.4 User Manual. *MRC Biostatistics Unit, Cambridge, UK,* Spiegelhalter, D.J., Best, N.G., Carlin, B.P. & Van Der Linde, A. (2002) Bayesian measures of model complexity and fit. *Journal of the Royal Statistical Society: Series B (Statistical Methodology),* **64,** 583-639.

Stewart-Oaten, A., Murdoch, W.W. & Parker, K.R. (1986) Environmental impact assessment:" Pseudoreplication" in time? *Ecology*, **67,** 929-940.

Sturtz, S., Ligges, U. & Gelman, A.E. (2005) R2WinBUGS: a package for running WinBUGS from R. *Journal of Statistical software*, **12,** 1-16.

Szerlag-Egger, S. & McRobert, S.P. (2007) Northern diamondback terrapin occurrence, movement, and nesting activity along a salt marsh access road. *Chelonian Conservation and Biology*, **6,** 295-301.

Table B1. Deviance Information Criterion (DIC) and ΔDIC for Robust Design candidate models varying capture (*p*) and site fidelity (γ) for a population of diamondback terrapins on Jekyll Island, Georgia, USA.

Parameter	**Model**	**DIC**	**ΔDIC[a]**
Capture (*p*)	sex + day + tide	4690.7	
	sex + day	4709.6	18.9
	sex + tide	4731.8	41.1
	sex	4957.0	266.3
Site fidelity (γ)	sex + year (fixed)	4690.7	
	sex	4700.6	9.9
	sex + year (random)	4762.3	71.6

[a] ΔDIC = difference in DIC from the best model.

Table B2. Mean and 95% Bayesian credibility intervals (CRI) of additional demographic rates of diamondback terrapins and hyperparameters obtained from the integrated model using two mark-recapture datasets.

Parameter	**Mean**	**95% CRI**[a]
Survival		
σ_y (male)	0.867	(0.025, 3.368)
σ_y (female)	0.551	(0.039, 1.777)
β_{cross}	1.521	(0.890, 2.367)
$\beta_{per.nocross}$	0.079	(-0.995, 1.215)
$\beta_{per.cross}$	0.684	(0.149, 1.453)
Detection		
p (male)	0.076	(0.048, 0.110)
p (female)	0.098	(0.052, 0.166)
σ_y	0.483	(0.257, 0.838)
p.c	0.746	(0.413, 0.988)
p.nc	0.055	(0.038, 0.080)
r	0.355	(0.211, 0.570)
Inclusion		
Ψ_{RD}	0.492	(0.396, 0.605)
Ψ_{MS}	0.543	(0.481, 0.607)

[a] 95% CRI: Bayesian credibility interval from the 2.5 to 97.5 percentile of the posterior distribution of parameter estimates

Table B3. Mean (± 90% confidence interval) estimated effects of management strategies on demographic rates, as obtained from empirical research (proportion of crossers in managed area, effects of hybrid barriers and flashing signage) and expert opinion, used in a population viability analysis for diamondback terrapins on Jekyll Island, GA.

Strategy	Proportion of crossers in managed area	Proportion blocked by barriers	Increase to crossing female survival	Nest success (Open)	Nest success (Hedges)	Nest success (Boxes)
Baseline	-	-	-	0.548 (0.448, 0.638)	0.156 (0.056, 0.246)	-
1 (SQ)	0.057	0.039 (0.013, 0.051)	-	0.548 (0.448, 0.638)	0.156 (0.056, 0.246)	0.963 (0.863, 0.966)
2	0.057	0.039 (0.013, 0.051)	-	0.548 (0.448, 0.638)	0.156 (0.056, 0.246)	0.963 (0.863, 0.966)
3	1.000	0.804 (0.655, 0.940)	-	0.874 (0.780, 0.945)	0.482 (0.388, 0.553)	0.975 (0.908, 0.977)
4	1.000	0.705 (0.586, 0.811)	-	0.518 (0.413, 0.600)	0.126 (0.021, 0.208)	0.962 (0.858, 0.965)
5	0.570	0.293 (0.210, 0.393)	-	0.874 (0.780, 0.945)	0.482 (0.388, 0.553)	0.975 (0.908, 0.977)
6	1.000	0.554 (0.413, 0.766)	0.035 (0.009, 0.056)	0.518 (0.413, 0.600)	0.126 (0.021, 0.208)	0.962 (0.858, 0.965)
7	0.057	0.039 (0.013, 0.051)	0.025 (0.005, 0.042)	0.548 (0.448, 0.638)	0.156 (0.056, 0.246)	0.963 (0.863, 0.966)
8	0.570	0.191 (0.125, 0.252)	0.025 (0.005, 0.042)	0.864 (0.770, 0.935)	0.472 (0.378, 0.543)	0.975 (0.907, 0.976)
9	0.057	0.039 (0.013, 0.051)	-	0.548 (0.448, 0.638)	0.156 (0.056, 0.246)	0.963 (0.863, 0.966)
10	0.399	0.218 (0.155, 0.300)	0.035 (0.009, 0.056)	0.719 (0.605, 0.805)	0.327 (0.213, 0.413)	0.969 (0.885, 0.972)
11	0.399	0.181 (0.123, 0.277)	0.035 (0.009, 0.056)	0.571 (0.469, 0.649)	0.179 (0.077, 0.257)	0.964 (0.866, 0.967)
12	0.317	0.194	0.035	0.491	0.099	0.961

		(0.141, 0.264)	(0.009, 0.056)	(0.391, 0.595)	(0.021, 0.203)	(0.855, 0.965)
13	0.431	0.282	-	0.759	0.367	0.971
		(0.230, 0.329)		(0.645, 0.845)	(0.253, 0.453)	(0.890, 0.973)
14	0.354	0.249	0.025	0.782	0.390	0.972
		(0.208, 0.287)	(0.005, 0.042)	(0.691, 0.868)	(0.299, 0.476)	(0.896, 0.974)
15	0.610	0.406	-	0.511	0.119	0.962
		(0.321, 0.500)		(0.411, 0.615)	(0.021, 0.223)	(0.858, 0.966)
16	0.657	0.463	0.025	0.759	0.367	0.971
		(0.359, 0.589)	(0.005, 0.042)	(0.645, 0.845)	(0.253, 0.453)	(0.890, 0.973)
17	0.657	0.463	0.025	0.800	0.408	0.972
		(0.359, 0.589)	(0.005, 0.042)	(0.703, 0.887)	(0.311, 0.495)	(0.898, 0.975)
18	0.735	0.429	-	0.501	0.109	0.961
		(0.328, 0.569)		(0.401, 0.601)	(0.021, 0.209)	(0.857, 0.965)
19	0.766	0.497	-	0.751	0.359	0.970
		(0.383, 0.634)		(0.634, 0.842)	(0.242, 0.450)	(0.888, 0.973)
20	1.000	0.586	-	0.795	0.403	0.972
		(0.435, 0.808)		(0.698, 0.882)	(0.306, 0.490)	(0.897, 0.975)

APPENDIX C

CODE FOR CHAPTER 4

The following code was run through R and JAGS to perform the integrated estimation models and population viability analyses.

Start of code:

```
#----------------------------------------------------------------------------------------------------------------#
##########Estimation phase: integrated model of two mark-recapture datasets#############
#----------------------------------------------------------------------------------------------------------------#
library(lattice)
library(coda)
library(R2jags)

#--------------------------------------------------#
###Import data and specify robust sampling design###
#--------------------------------------------------#
terpRD <- read.csv("TerpRD.csv")   #capture history data from seining (2010-2015)
cov <- read.csv("Tide_temp_cov.csv")   #Tide covariates for sampling occasions
z.data <- read.csv("z_known.csv") #known inclusion states based on capture histories
f.data <- read.csv("f_known.csv") #known fidelity (availability) states based on capture histories

#Sampling information and covariates
n1=8 #n1 = number of primary periods
n2=18 #n2 = number of total secondary sampling periods
nss=c(0,0,3,3,3,3,3,3) # nss[i] = number of secondary periods in primary period i
cnss=c(0,0,0,3,6,9,12,15) # cnss[i] = cumulative sum of nss from 1 to i-1
tide=cov$cov_tide
terpRD <- terpRD[,1:21]
sign <- c(-1,-1,-1,-1,-1,1,1,1)

#Augment dataset and produce data file for individual sex (sex) and capture histories (yaug)
nz<-5*250          #sets number of augmented '0' rows to 4 times the number of observed
individuals
aug<-array(0,dim=c(nz,length(terpRD[1,])))
```

```
aug<-as.data.frame(aug)
names(aug)<-names(terpRD)
yaug.rd<-rbind(terpRD,aug)
sex<-yaug.rd[,1]     #0 = male, 1 = female
yaug.rd<-yaug.rd[,4:21]

aug.sub<-as.matrix(aug[,4:11])
z.known<-as.matrix(z.data[,1:8])
z.known<-rbind(z.known,aug.sub)
z.known[z.known==0]<-NA

z.inits <- z.known
z.inits[is.na(z.inits)]<-1
z.inits[z.known==1]<-NA
z.inits[,1]<-NA
z.inits[,2]<-1

f.known<-as.matrix(f.data[,1:8])
f.known<-rbind(f.known,aug.sub)
f.known[f.known==0]<-NA

f.inits <- f.known
f.inits[is.na(f.inits)]<-1
f.inits[f.known==1]<-NA
f.inits[,1:2]<-NA

w.rd.inits <- rep(1, nrow(yaug.rd))

#-------------------------------------------------------#
###Import data for multistate CR study---------###
#-------------------------------------------------------#
terpLD <- read.csv("Terp_LD_4states.csv")   #the data

#Augment dataset and produce data file for individual sex (sex) and capture histories (yaug)
nz<-4*length(terpLD[,1])          #sets number of augmented '0' rows to 4 times the number of
observed individuals
aug<-array(4,dim=c(nz,length(terpLD[1,])))
aug<-as.data.frame(aug)
names(aug)<-names(terpLD)
yaug.ld<-rbind(terpLD,aug)

#Create z known data
z.ld.known<-as.matrix(yaug.ld)
z.ld.known[z.ld.known==4]<-NA
z.ld.known[z.ld.known==3]<-5
z.ld.known[z.ld.known==2]<-3
```

```
z.ld.known[z.ld.known==1]<-2

#Compute year of known death and refine z.ld.known states to be fixed in dead state after
observed dead on road
d <- which(z.ld.known==5,arr.ind=T)
for (i in 1:nrow(d)){
  ifelse(d[i,2]!=ncol(z.ld.known), z.ld.known[d[i,1], (d[i,2]+1):ncol(z.ld.known)] <- 6, next)}

#Create good initial values for z
z.ld.inits <- z.ld.known
z.ld.inits[is.na(z.ld.inits)]<-1
z.ld.inits[z.ld.known==2]<-9
z.ld.inits[z.ld.known==3]<-9
z.ld.inits[z.ld.known==5]<-9
z.ld.inits[z.ld.known==6]<-9

first <- array(NA,dim=nrow(z.ld.inits))
for (i in 1:nrow(z.ld.inits)){          #Function creates vector of occasion of first observation, and
replaces any unobserved states after first observation to 4
  ifelse(sum(z.ld.inits[i,])!=8,first[i] <- min(which(z.ld.inits[i,]==9)), next)}
first[is.na(first)]<-8
for (i in 1:nrow(z.ld.inits)){
  for (t in (first[i]+1):ncol(z.ld.inits)){
    ifelse(first[i]!=8,
           ifelse(z.ld.inits[i,t]!=9, z.ld.inits[i,t] <- 4, next),next)}}
z.ld.inits[z.ld.inits==9]<-NA
z.ld.inits[,1]<-NA

w.ld.inits <- rep(1, nrow(yaug.ld))

#----------------------------------------------------#
###The model: Robust design & Multistate###
#----------------------------------------------------#

sink("ipm.jags")
cat("
    model{
    # RD model-----------------------------------
    # Survival: Fixed effect for group (M/F) with year factor pre and post sign mgmt (2010-
2012/2013-2014) and random year effect
    # Fidelity (gamma): fixed group, random year effects
    # Capture: Fixed effect for group, random temporal effect, and fixed effect of tide amplitude
    # Recapture: = capture
    # Inclusion (omega): Constant

    #LD model------------------------------------
```

```
# Parameters:
# s.c: survival probability of crossing terrapins
# s.nc: survival probability of non-crossing terrapins
# b: entry probability
# gamma: fidelity (1-temporary emigration) probability
# c: crossing status fidelity
# r: recovery probability
# p: capture probability
#
# States (S):
# 1 not yet entered
# 2 alive crossing
# 3 alive not crossing
# 4 alive unavailable
# 5 dead crossing
# 6 dead not crossing
#
# Observations (O):
# 1 seen alive crossing
# 2 seen alive not crossing
# 3 recovered dead
# 4 not seen or recovered
#-------------------------------------

#-------------
#Section 1. Define priors for all parameters#
#-------------
#---------Shared parameters: survival, entry, and temporary emigration

#Survival parameters
for (i in 1:M) {
g[i] <- sex[i]+1
for (y in 1:(n1-1)){
phi[i,y] <- phi.g[g[i],y]
} #y
} #i

for (u in 1:2) {   #RD data with two sexes
for (y in 1:(n1-1)){
phi.eta.rd[u,y] <- phi.mu[u] + delta[u]*sign[y] + phi.eps[u,y]
phi.g[u,y] <- 1/(1+exp(-phi.eta.rd[u,y]))
phi.eps[u,y] ~ dnorm(0,phi.tau[u])
} #y
phi.mean[u] ~ dunif(0,1)                  #Prior for mean survival
phi.mu[u] <- log(phi.mean[u]/(1-phi.mean[u]))   #logit transformation
phi.tau[u] <- pow(phi.sd[u],-2)
```

```
   phi.sd[u] ~ dunif(0,5)                    #Prior for sd survival
   } #u

   for (u in 1:2){
   for (y in 1:(n1-1)){                  # LD data with females only
   phi.eta.ld[u,y] <- phi.mu[2] + phi.eps[2,y] + delta[2]*sign[y] + cr.effect[u] + cr.int[u]*sign[y]
#Crossing effect and interaction on survival
   } #y
   } #u
   for (y in 1:(n1-1)){
   s.nc[y] <- 1/(1+exp(-phi.eta.ld[1,y]))     # Prior for survival of non-crossing terps
   s.c[y] <- 1/(1+exp(-phi.eta.ld[2,y]))       # Prior for survival of crossing terps
   } #y

   delta[1] <- 0
   delta[2] ~ dnorm(0,0.37)    #Prior for fixed effect for managed (sign) years
   cr.effect[1] ~ dnorm(0,0.37)   #Prior for effect of not crossing on survival
   cr.effect[2] <- -cr.effect[1]  #Prior for effect of crossing on survival
   cr.int[1] ~ dnorm(0,0.37)   #Prior for interaction effect of not crossing on survival
   cr.int[2] <- -cr.int[1]  #Prior for interaction effect of crossing on survival

   #Entry probabilities

   for (i in 1:M) {
   for (y in 1:(n1-1)) {
   nu[i,y] <- nu.g[g[i],y]     #Use nu for RD model
   } #y
   } #i

   for (u in 1:2) {
   for (y in 1:(n1-1)) {
   ent[u,y] ~ dgamma(1, 1)    # Dirichlet prior for entry probability
   b[u,y] <- ent[u,y] / sum(ent[u,1:(n1-1)])
   } #y
   } #u

   for (u in 1:2) {
   nu.g[u,1] <- b[u,1]    #Use nu.g[2,] for LD model
   for (y in 2:(n1-1)) {
   nu.g[u,y] <- b[u,y] / (1-sum(b[u,1:(y-1)]))
   } #y
   } #u

   #Temporary emigration

   for (i in 1:M) {
```

```
   for (y in 1:(n1-1)){
   gamma[i,y] <- gamma.g[g[i],y]
   } #t
   } #i

   for (u in 1:2) {
   for (y in 1:(n1-1)){
   gamma.eta[u,y] <- gam.mu[u] + gam.eps[u,y]
   gamma.g[u,y] <- 1/(1+exp(-gamma.eta[u,y]))
   gam.eps[u,y] ~ dnorm(0,gam.tau[u])
   } #y
   gam.mean[u] ~ dunif(0,1)                   #Prior for mean fidelity
   gam.mu[u] <- log(gam.mean[u]/(1-gam.mean[u]))   #logit transformation
   gam.tau[u] <- pow(gam.sd[u],-2)
   gam.sd[u] ~ dunif(0,5)                     #Prior for sd fidelity
   } #u

   #---------Parameters of independent models

   #for RD model
   #Capture parameters
   for (i in 1:M) {
   for (t in 1:n2) {
   p[i,t] <- p.g[g[i],t]
   } #t
   } #i

   for (u in 1:2){
   for (t in 1:n2) {
   p.eta[u,t] <- p.mu[u] + alpha*tide[t] + p.eps[t]   #capture with fixed group and fixed covariate
effects
   p.g[u,t] <- 1/(1+exp(-p.eta[u,t]))        #back-transformed group and year-specific capture
   } #t
   p.mean[u] ~ dunif(0,1)                   #Prior for mean survival
   p.mu[u] <- log(p.mean[u]/(1-p.mean[u]))   #logit transformation
   } #u
   for (t in 1:n2) {
   p.eps[t] ~ dnorm(0,p.tau)
   } #t
   p.tau <- pow(p.sd,-2)
   p.sd ~ dunif(0,5)
   alpha ~ dnorm(0,0.37)        # Fixed effect for daily tide amplitude
   psi.rd ~ dunif(0,1)    #Prior for latent inclusion

   #for LD model
   for (y in 1:(n1-1)){
```

```
   p.c[y] <- mean.p.c
   p.nc[y] <- mean.p.nc
   c[y] <- mean.c
   r[y] <- mean.r
   } #y

   mean.p.c ~ dunif(0, 1)    # Prior for mean capture
   mean.p.nc ~ dunif(0, 1)   # Prior for mean capture
   mean.c ~ dunif(0, 1)      # Prior for mean crossing status fidelity
   mean.r ~ dunif(0,1)       # Prior for mean recovery probability
   psi.ld ~ dunif(0, 1)      # Prior for latent inclusion

   #---------Define state-transition and observation matrices for live-dead model
   for (i in 1:nind){
   # Define probabilities of state S(y+1) given S(y)
   for (y in 1:(n1-1)){

   ps[1,i,y,1] <- 1-nu.g[2,y]
   ps[1,i,y,2] <- nu.g[2,y] * gamma.g[2,y] * c[y] * s.c[y]
   ps[1,i,y,3] <- nu.g[2,y] * gamma.g[2,y] * (1-c[y]) * s.nc[y]
   ps[1,i,y,4] <- nu.g[2,y] * (1-gamma.g[2,y]) * s.nc[y]
   ps[1,i,y,5] <- nu.g[2,y] * gamma.g[2,y] * c[y] * (1-s.c[y])
   ps[1,i,y,6] <- nu.g[2,y] * (1-gamma.g[2,y]) * (1-s.nc[y]) + nu.g[2,y] * gamma.g[2,y] * (1-c[y])
* (1-s.nc[y])
   ps[2,i,y,1] <- 0
   ps[2,i,y,2] <- gamma.g[2,y] * c[y] * s.c[y]
   ps[2,i,y,3] <- gamma.g[2,y] * (1-c[y]) * s.nc[y]
   ps[2,i,y,4] <- (1-gamma.g[2,y]) * s.nc[y]
   ps[2,i,y,5] <- gamma.g[2,y] * c[y] * (1-s.c[y])
   ps[2,i,y,6] <- (1-gamma.g[2,y]) * (1-s.nc[y]) + gamma.g[2,y] * (1-c[y]) * (1-s.nc[y])
   ps[3,i,y,1] <- 0
   ps[3,i,y,2] <- gamma.g[2,y] * c[y] * s.c[y]
   ps[3,i,y,3] <- gamma.g[2,y] * (1-c[y]) * s.nc[y]
   ps[3,i,y,4] <- (1-gamma.g[2,y]) * s.nc[y]
   ps[3,i,y,5] <- gamma.g[2,y] * c[y] * (1-s.c[y])
   ps[3,i,y,6] <- (1-gamma.g[2,y]) * (1-s.nc[y]) + gamma.g[2,y] * (1-c[y]) * (1-s.nc[y])
   ps[4,i,y,1] <- 0
   ps[4,i,y,2] <- gamma.g[2,y] * c[y] * s.c[y]
   ps[4,i,y,3] <- gamma.g[2,y] * (1-c[y]) * s.nc[y]
   ps[4,i,y,4] <- (1-gamma.g[2,y]) * s.nc[y]
   ps[4,i,y,5] <- gamma.g[2,y] * (1-s.c[y])
   ps[4,i,y,6] <- (1-gamma.g[2,y]) * (1-s.nc[y]) + gamma.g[2,y] * (1-c[y]) * (1-s.nc[y])
   ps[5,i,y,1] <- 0
   ps[5,i,y,2] <- 0
   ps[5,i,y,3] <- 0
   ps[5,i,y,4] <- 0
```

```
    ps[5,i,y,5] <- 0
    ps[5,i,y,6] <- 1
    ps[6,i,y,1] <- 0
    ps[6,i,y,2] <- 0
    ps[6,i,y,3] <- 0
    ps[6,i,y,4] <- 0
    ps[6,i,y,5] <- 0
    ps[6,i,y,6] <- 1

    # Define probabilities of O(y) given S(y)
    po[1,i,y,1] <- 0
    po[1,i,y,2] <- 0
    po[1,i,y,3] <- 0
    po[1,i,y,4] <- 1
    po[2,i,y,1] <- p.c[y]
    po[2,i,y,2] <- 0
    po[2,i,y,3] <- 0
    po[2,i,y,4] <- 1-p.c[y]
    po[3,i,y,1] <- 0
    po[3,i,y,2] <- p.nc[y]
    po[3,i,y,3] <- 0
    po[3,i,y,4] <- 1-p.nc[y]
    po[4,i,y,1] <- 0
    po[4,i,y,2] <- 0
    po[4,i,y,3] <- 0
    po[4,i,y,4] <- 1
    po[5,i,y,1] <- 0
    po[5,i,y,2] <- 0
    po[5,i,y,3] <- r[y]
    po[5,i,y,4] <- 1-r[y]
    po[6,i,y,1] <- 0
    po[6,i,y,2] <- 0
    po[6,i,y,3] <- 0
    po[6,i,y,4] <- 1
    } #y
    } #i

    #-------------
    #Section 2. Likelihoods of single data sets#
    #-------------

    #--------Robust design part
    # State process #
    for (i in 1:M){
    w.rd[i] ~ dbern(psi.rd)            # Inclusion parameter representing individual is real and not
fake augmented row
```

```
z.rd[i,1] <- 0               # Make sure all individuals are in state 0 (not entered) at time 1
f[i,1] <- 0                  # However, availability is set to 0 in years before creek sampling started
q[i,1] <- 1-z.rd[i,1]                        # Availability for recruitment
mu1a[i,2] <- phi[i,1] * z.rd[i,1]            # Prob surviving to t given alive at t-1
mu1b[i,2] <- nu[i,1] * q[i,1]                # Prob entering superpop given available
mu1[i,2] <- mu1a[i,2] + mu1b[i,2]
z.rd[i,2] ~ dbern(mu1[i,2])
f[i,2] <- 0

# Subsequent year/occasion, state process
for (y in 3:n1) {      #Loop over primary periods (years)
# State process
q[i,y-1] <- 1-z.rd[i,y-1]                        # Availability for recruitment
mu1a[i,y] <- phi[i,y-1] * z.rd[i,y-1]            # Prob surviving to t given alive at t-1
mu1b[i,y] <- nu[i,y-1] * prod(q[i,1:(y-1)])            # Prob entering superpop given available
mu1[i,y] <- mu1a[i,y] + mu1b[i,y]
z.rd[i,y] ~ dbern(mu1[i,y])
avail[i,y] <- gamma[i,y-1] * z.rd[i,y]    #changed
f[i,y] ~ dbern(avail[i,y])

# Observation process
for (j in 1:nss[y]) {   #Loop over secondary periods (times / days)
y.rd[i,(cnss[y]+j)] ~ dbern(p.eff[i,(cnss[y]+j)])
p.eff[i,(cnss[y]+j)] <- f[i,y] * p[i,(cnss[y]+j)] * w.rd[i]

} #j secondary
} #y primary
} #i individual

#--------Live-dead part
# Likelihood
for (i in 1:nind){
# Define latent state at first occasion
w.ld[i] ~ dbern(psi.ld)   #Draw latent inclusion
z.ld[i,1] <- 1         #Make sure all individuals are in state 1 (not entered) at time 1
for (y in 2:n1){
# State process: draw S(y) given S(y-1)
z.ld[i,y] ~ dcat(ps[z.ld[i,y-1], i, y-1,])
# Observation process: draw O(y) given S(y)
z.real[i,y] <- z.ld[i,y]*w.ld[i] + 4*(1-w.ld[i])
y.ld[i,y] ~ dcat(po[z.real[i,y], i, y-1,])
} #t
} #i

#---------
#Section 3. Derived parameters#
```

```
#--------from RD model

#Abundance
for (i in 1:M) {
for (t in 1:n1) {
u.rd[i,t] <- z.rd[i,t] * w.rd[i]    #Deflated latent state (u)
} #t
} #i

for (t in 1:n1) {
N.loc[t] <- sum(u.rd[1:M,t])                    # Annual superpop size
N.loc.fem[t] <- inprod(sex[1:M], u.rd[1:M,t])      # Inner product operator -- sums of products of 2 vectors
N.loc.male[t] <- N.loc[t] - N.loc.fem[t]
} #t
for (i in 1:M) {
N.loc.ind[i] <- sum(u.rd[i, 2:n1])
N.loc.alive[i] <- 1-equals(N.loc.ind[i], 0)
} #i

Nsuper.loc <- sum(N.loc.alive[])                              # Superpopulation size

# From LD model: Calculate derived population parameters

for (i in 1:nind){
for (t in 1:n1){
u.ld[i,t] <- z.ld[i,t] * w.ld[i]    #Deflated latent state (u)
} #t
} #i

for (i in 1:nind){
for (t in 2:n1){
al.c[i,t-1] <- equals(u.ld[i,t], 2)
al.nc[i,t-1] <- equals(u.ld[i,t], 3)
al.out[i,t-1] <- equals(u.ld[i,t], 4)
d.c[i,t-1] <- equals(u.ld[i,t], 5)
d.out[i,t-1] <- equals(u.ld[i,t], 6)
} #t
real[i] <- sum(al.c[i,]) + sum(al.nc[i,]) + sum(d.c[i,]) + sum(al.out[i,]) + sum(d.out[i,])
} #i

for (t in 1:(n1-1)){
N.out[t] <- sum(al.out[,t])
N.cross[t] <- sum(al.c[,t])
N.nocross[t] <- sum(al.nc[,t])
N.dead[t] <- sum(d.c[,t])
```

```
    } #t

    for (i in 1:nind){
    ind[i] <- 1-equals(real[i],0)
    } #i

    Nsuper.JIC <- sum(ind[])          # Superpopulation size

    } #END OF MODEL
    ",fill=TRUE)
sink()

#Bundle data
ipm.data<-
list(y.rd=yaug.rd,sex=sex,M=nrow(yaug.rd),n1=n1,n2=n2,nss=nss,cnss=cnss,tide=tide,z.rd=z.kn
own,f=f.known,sign=sign, y.ld=yaug.ld, nind=nrow(yaug.ld), z.ld=z.ld.known)

#Parameters monitored
params<-
c("phi.mean","phi.sd","s.c","s.nc","delta","cr.effect","cr.int","p.mean","p.sd","mean.p.c","mean.
p.nc","mean.c","mean.r","nu.g","gam.mean","gam.sd","psi.rd","psi.ld","N.loc.fem","N.loc.male"
,"Nsuper.loc","Nsuper.JIC","N.out","N.cross","N.nocross","N.dead")

#MCMC settings
ni<-60000
nt<-1
nb<-50000
nc<-1

###################   PARALLELIZE !!! #######################
library(foreach)
library(doParallel)
library(R2jags); load.module("glm"); load.module("lecuyer")

init.fun <- function() {
  list(.RNG.name = "lecuyer::RngStream",
       .RNG.seed = runif(1,1,1000000),
       phi.mean=runif(2,0.5,1),phi.sd=runif(2,0,5),delta=c(NA,
rnorm(1)),cr.effect=c(rnorm(1),NA),cr.int=c(rnorm(1),NA),

gam.mean=runif(2,0,1),gam.sd=runif(2,0,5),p.mean=runif(2,0,1),p.sd=runif(1,0,5),alpha=rnorm(
1),psi.rd=runif(1,0,1),
       z.rd=z.inits,f=f.inits,w.rd=w.rd.inits,mean.p.c=runif(1,0,1),mean.p.nc=runif(1,0,1),
       mean.c=runif(1,0,1),mean.r=runif(1,0,1),z.ld=z.ld.inits,w.ld=w.ld.inits)    ##  Here are my
initial values.
}
```

```
registerDoParallel(cl = 3, cores = 3)          ##  Reserve the cores...

mod.time <- system.time(
  samp <- foreach(i=1:getDoParWorkers(),
                  .export = c("jags", "as.mcmc.list", "load.module"),       ###  needed functions in
{r2jags} need to be registered here
                  .combine = "c",                ###  tells foreach() how to put the different output objects
together.  Since coda.samples() creates an object of class "mcmc.list", using c() on the result
objects (of which there are four in this case), will create a new list with four elements, that
happens to have the right structure to serve as an "mcmc.list" object with four chains.
Demonstration: try "L <- c(list(1:3), list(letters()))", then "class(L)".
                  .final = mcmc.list,
                  .verbose = T) %dopar% {
                    load.module("lecuyer")
                    load.module("dic")
                    IPM.LWBD <- jags(data=ipm.data,inits=init.fun,params," ipm.jags",
n.chains=nc,n.thin=nt,n.iter=ni,n.burnin=nb,
                                     working.directory=getwd())
                    result <- as.mcmc.list(IPM.LWBD$BUGSoutput)
                    return(result)        ###  each core will produce an object called "result", of class
"mcmc.list"
                  }
  )             ###
######################   END PARALLELIZE !!! ##########################

IPM.jagsout <- rbind(samp[[1]],samp[[2]],samp[[3]])
write.csv(IPM.jagsout,"C:/…/IPM.results.csv")    #Save MCMC posterior samples to disk
```

```
#-----------------------------------------------------------------------------------------------------------------#
##########Prediction phase: stage-based PVA model for diamondback terrapins###########
#-----------------------------------------------------------------------------------------------------------------#
#-------------------------------------------------#
###Setup population model space###
#-------------------------------------------------#
#Load parameter estimates and management effects from files
IPM <- read.csv("IPM.csv")     # Posterior parameter estimates (MCMC samples) from IPM
mgmt.effects <- read.csv("Management effects.csv")   # Mgmt effects for each alternative strategy

#Specify params from mgmt effects table
p.mgmt <- mgmt.effects[,2]
p.block <- mgmt.effects[,3]
p.block.LCI <- mgmt.effects[,4]
p.block.UCI <- mgmt.effects[,5]
s.cross.eff <- mgmt.effects[,6]
s.cross.eff <- log(s.cross.eff/(1-s.cross.eff))
s.cross.eff.LCI <- mgmt.effects[,7]
s.cross.eff.LCI <- log(s.cross.eff.LCI/(1-s.cross.eff.LCI))
s.cross.eff.UCI <- mgmt.effects[,8]
s.cross.eff.UCI <- log(s.cross.eff.UCI/(1-s.cross.eff.UCI))
p.hab <- mgmt.effects[,12:14]
s.nest <- mgmt.effects[,15:17]
s.nest.LCI <- mgmt.effects[,18:20]
s.nest.UCI <- mgmt.effects[,21:23]

#Specify PVA environmnet (iterations, years, scenarios)
iter = 1000                              # Number of iterations per scenario/strategy
n.yrs = 50                               # Number of years to project pop for each iteration
strat <- dim(mgmt.effects)[1]                     # Number of strategies to cycle through

#Create blank containers
##(Adult survival and stochasticity)
s.M.ad<-array(dim=c(iter))
s.F.ad<-array(dim=c(iter))
s.M.ad.sd<-array(dim=c(iter))
s.F.ad.sd<-array(dim=c(iter))
s.M.ad.mu<-array(dim=c(iter))
s.F.ad.mu<-array(dim=c(iter))
cr.effect.1<-array(dim=c(iter))
cr.effect.2<-array(dim=c(iter))
delta.1<-array(dim=c(iter))
delta.2<-array(dim=c(iter))
s.M.yr.var<-array(dim=c(n.yrs,iter))
s.F.yr.var<-array(dim=c(n.yrs,iter))
```

```
s.M.ad.yr<-array(dim=c(n.yrs,iter,strat))
s.F.cross<-array(dim=c(n.yrs,iter,strat))
s.F.nocross<-array(dim=c(n.yrs,iter,strat))
##(Recruitment [juvenile graduation prob])
g.M.juv1<-array(dim=c(n.yrs,iter))
g.F.juv1<-array(dim=c(n.yrs,iter))
##(Remaining parameters for stage-based matrix)
gam<-array(dim=c(n.yrs,iter))
p.cross<-array(dim=c(n.yrs,iter))
M.juv<-array(dim=c(n.yrs+1,3,iter,strat))
F.juv<-array(dim=c(n.yrs+1,3,iter,strat))
N.nester<-array(dim=c(n.yrs+1,iter,strat))
N.nester.hab<-array(dim=c(n.yrs+1,3,iter,strat))
N.nest.hab<-array(dim=c(n.yrs+1,3,iter,strat))
N.nest.surv.hab<-array(dim=c(n.yrs+1,3,iter,strat))
N.eggs.hab<-array(dim=c(n.yrs+1,3,iter,strat))
N.eggs.surv.hab<-array(dim=c(n.yrs+1,3,iter,strat))
N.M.new<-array(dim=c(n.yrs+1,3,iter,strat))
N.F.new<-array(dim=c(n.yrs+1,3,iter,strat))
N.F.start<-array(dim=c(n.yrs+1,iter,strat))
N.F.cat<-array(dim=c(n.yrs+1,3,iter,strat))
N.mgmt<-array(dim=c(n.yrs+1,iter,strat))
N.block<-array(dim=c(n.yrs+1,iter,strat))
N.cross<-array(dim=c(n.yrs+1,iter,strat))
N.nocross<-array(dim=c(n.yrs+1,iter,strat))
N.surv.cross<-array(dim=c(n.yrs+1,iter,strat))
N.surv.nocross<-array(dim=c(n.yrs+1,iter,strat))
N.surv.nouser<-array(dim=c(n.yrs+1,iter,strat))
##(Population growth rate and persistence)
lambda<-array(dim=c(n.yrs,iter,strat))
crash <- array(dim=c(iter,strat))
mean.lambda.iter<-array(dim=c(iter,strat))
mean.lambda.mgmt<-array(dim=strat)
perc.change.iter<-array(dim=c(iter,strat))
perc.change.mgmt<-array(dim=strat)
persist<-array(dim=c(iter,strat))
persistence<-array(dim=c(iter,strat))
persistence.prob<-array(dim=strat)

#Derive params from IPM estimates
#-abundance
N.loc.fem.avg <-
mean(c(IPM$N.loc.fem.2,IPM$N.loc.fem.3,IPM$N.loc.fem.4,IPM$N.loc.fem.5,IPM$N.loc.fem.
6,IPM$N.loc.fem.7,IPM$N.loc.fem.8))
```

```
N.loc.male.avg <-
mean(c(IPM$N.loc.fem.2,IPM$N.loc.male.3,IPM$N.loc.male.4,IPM$N.loc.male.5,IPM$N.loc.
male.6,IPM$N.loc.male.7,IPM$N.loc.male.8))
sex.ratio <- N.loc.fem.avg/N.loc.male.avg
N.JIC.fem <- cbind(IPM$N.cross.1+IPM$N.nocross.1+IPM$N.out.1,
               IPM$N.cross.2+IPM$N.nocross.2+IPM$N.out.2,
               IPM$N.cross.3+IPM$N.nocross.3+IPM$N.out.3,
               IPM$N.cross.4+IPM$N.nocross.4+IPM$N.out.4,
               IPM$N.cross.5+IPM$N.nocross.5+IPM$N.out.5,
               IPM$N.cross.6+IPM$N.nocross.6+IPM$N.out.6,
               IPM$N.cross.7+IPM$N.nocross.7+IPM$N.out.7)
N.JIC.fem <- round(mean(N.JIC.fem),0)
N.JIC.male <- round(N.JIC.fem*(1/sex.ratio),0)

# survival
#Simulate stochastic demographic survival rates from IPM estimates
for (i in 1:iter){
  row[i] <- sample(row(IPM),1,replace=T)
  s.M.ad[i] <- IPM$phi.mean.1[row[i]]
  s.F.ad[i] <- IPM$phi.mean.2[row[i]]
  s.M.ad.sd[i] <- IPM$phi.sd.1[row[i]]
  s.F.ad.sd[i] <- IPM$phi.sd.2[row[i]]
  cr.effect.1[i] <- IPM$cr.effect.1[row[i]]
  cr.effect.2[i] <- IPM$cr.effect.2[row[i]]
  delta.1[i] <- IPM$delta.1[row[i]]
  delta.2[i] <- IPM$delta.2[row[i]]
} #i
for (i in 1:iter){
  s.M.ad[i] <- sample(IPM$phi.mean.1,1,replace=T)
  s.F.ad[i] <- sample(IPM$phi.mean.2,1,replace=T)
  s.M.ad.sd[i] <- sample(IPM$phi.sd.1,1,replace=T)
  s.F.ad.sd[i] <- sample(IPM$phi.sd.2,1,replace=T)
  cr.effect.1[i] <- sample(IPM$cr.effect.1,1,replace=T)
  cr.effect.2[i] <- sample(IPM$cr.effect.2,1,replace=T)
  delta.1[i] <- sample(IPM$delta.1,1,replace=T)
  delta.2[i] <- sample(IPM$delta.2,1,replace=T)
} #i

for (t in 1:n.yrs){
  for (i in 1:iter){
    for (m in 1:strat){
      s.M.ad.mu[i] <- log(s.M.ad[i]/(1-s.M.ad[i]))
      s.M.yr.var[t,i] <- rnorm(1,0,s.M.ad.sd[i])
      s.M.ad.yr[t,i,m] <- 1/(1+exp(-(s.M.ad.mu[i] + s.M.yr.var[t,i])))
      s.F.ad.mu[i] <- log(s.F.ad[i]/(1-s.F.ad[i]))
      s.F.yr.var[t,i] <- rnorm(1,0,s.F.ad.sd[i])
```

```
    } #m
      s.F.cross[t,i,1] <- 1/(1+exp(-(s.F.ad.mu[i] + s.F.yr.var[t,i] - delta.2[i] + cr.effect.2[i] +
s.cross.eff[1])))
      s.F.nocross[t,i,1] <- 1/(1+exp(-(s.F.ad.mu[i] + s.F.yr.var[t,i] - delta.1[i] + cr.effect.1[i])))
    for (m in 2:strat){
      s.F.cross[t,i,m] <- 1/(1+exp(-(s.F.ad.mu[i] + s.F.yr.var[t,i] + cr.effect.2[i] + delta.2[i] +
s.cross.eff[m])))
      s.F.nocross[t,i,m] <- 1/(1+exp(-(s.F.ad.mu[i] + s.F.yr.var[t,i] + cr.effect.1[i] + delta.1[i])))
    } #m
  } #i
} #t

#-p.cross, gamma, and entry prob (recruitment) <- replaces graduation prob from juveniles to
adults
p.cross.yr <-
cbind((IPM$N.cross.1+IPM$N.dead.1)/(IPM$N.cross.1+IPM$N.dead.1+IPM$N.nocross.1),
                  (IPM$N.cross.2+IPM$N.dead.2)/(IPM$N.cross.2+IPM$N.dead.2+IPM$N.nocross.2),
                  (IPM$N.cross.3+IPM$N.dead.3)/(IPM$N.cross.3+IPM$N.dead.3+IPM$N.nocross.3),
                  (IPM$N.cross.4+IPM$N.dead.4)/(IPM$N.cross.4+IPM$N.dead.4+IPM$N.nocross.4),
                  (IPM$N.cross.5+IPM$N.dead.5)/(IPM$N.cross.5+IPM$N.dead.5+IPM$N.nocross.5),
                  (IPM$N.cross.6+IPM$N.dead.6)/(IPM$N.cross.6+IPM$N.dead.6+IPM$N.nocross.6),
                 (IPM$N.cross.7+IPM$N.dead.7)/(IPM$N.cross.7+IPM$N.dead.7+IPM$N.nocross.7))
p.cross.mean <- rowMeans(p.cross.yr)
nu.yr.M <-
cbind(IPM$nu.g.1.1,IPM$nu.g.1.2,IPM$nu.g.1.3,IPM$nu.g.1.4,IPM$nu.g.1.5,IPM$nu.g.1.6)
nu.mean.M <- rowMeans(nu.yr.M)
nu.yr.F <-
cbind(IPM$nu.g.2.1,IPM$nu.g.2.2,IPM$nu.g.2.3,IPM$nu.g.2.4,IPM$nu.g.2.5,IPM$nu.g.2.6)
nu.mean.F <- rowMeans(nu.yr.F)

for (t in 1:n.yrs){
  for (i in 1:iter){
    gam[t,i] <- sample(IPM$gam.mean.2,1,replace=T)
    p.cross[t,i] <- sample(p.cross.mean,1,replace=T)
  } #t
} #i
#Specify constant demographic rates from literature
s.M.hat <- s.F.hat <- 0.53
s.M.juv <- s.F.juv <- 0.57
p.M.juv <- 0.472268
g.M.juv <- 0.097732
p.F.juv <- 0.519252
g.F.juv <- 0.050748
s.M.av <- mean(s.M.ad)
s.F.av <- mean(s.F.ad)
clutch <- 2         #avg number of clutches per female per year
```

```
clutch.size <- 6.9  #avg number of eggs per clutch per female per year
s.egg <- 0.79
fert <- clutch*clutch.size*s.egg*0.39*(s.F.hat^.75)
fem.hatch.ratio <- c(1,0.15,1)
fem.hatch.ratio.LCI <- c(0.51,0,0.51)

#Calculate stable age distribution for initial population sizes
Lefkovitch <- c(0,      0,      0,      0,      0,      fert*0.5,
            s.M.hat, p.M.juv, 0,      0,      0,      0,
            0,      g.M.juv, s.M.av, 0,      0,      0,
            0,      0,      0,      0,      0,      fert*0.5,
            0,      0,      0,      s.F.hat, p.F.juv, 0,
            0,      0,      0,      0,      g.F.juv, s.F.av)
terp <- matrix(Lefkovitch,nrow=6,byrow=T)
terp_eigen <- eigen(terp)
stable_stage <- terp_eigen$vectors[,1]/sum(terp_eigen$vectors[,1])
stable_stage <-as.numeric(stable_stage)

Y.ad0<-N.JIC.male  #mean from model'
X.ad0<-N.JIC.fem  #mean from model'
Y.hat0<-round((Y.ad0/stable_stage[3])*stable_stage[1],0)
Y.juv0<-round((Y.ad0/stable_stage[3])*stable_stage[2],0)
X.hat0<-round((Y.ad0/stable_stage[3])*stable_stage[4],0)
X.juv0<-round((Y.ad0/stable_stage[3])*stable_stage[5],0)

#Derived stable stage distribution from previous runs
stage.prop <- c(0.16212451, 0.23016752, 0.13783620, 0.16212451, 0.26327821, 0.04446905)

#Initial abundances
Y.ad0<-N.JIC.male  #mean from model'
X.ad0<-N.JIC.fem  #mean from model'
Y.hat0<-round((Y.ad0/stage.prop[3])*stage.prop[1],0)
Y.juv0<-round((Y.ad0/stage.prop[3])*stage.prop[2],0)
X.hat0<-round((Y.ad0/stage.prop[3])*stage.prop[4],0)
X.juv0<-round((Y.ad0/stage.prop[3])*stage.prop[5],0)

Total0<-Y.hat0+Y.juv0+Y.ad0+X.hat0+X.juv0+X.ad0
N0 <- c(Y.hat0,Y.juv0,Y.ad0,X.hat0,X.juv0,X.ad0,Total0)
years <- c(1:(n.yrs+1))
stages <- c("Y.hat","Y.juv","Y.ad","X.hat","X.juv","X.ad","N.total")
dimnames <- list(years,stages)
N<-matrix(data=NA,nrow=n.yrs+1,ncol=length(N0),dimnames=dimnames)
N<-array(dim=c(length(N[,1]),length(N[1,]),iter,strat))
dim(N)   #Check if dimensions are right (1=years, 2=stages, 3=iterations, 4=mgmt scenarios)

######Sensitivity analysis section#####
```

```
#Parameters to replace current model parameters  (1-way)
#Parameters to replace current model parameters  (2-way combinations)
#gam, p.cross

#------------------------------------------------------------------------------------------------#
###Population Viability Analysis - stage-based model for 50 years, 21 management scenarios,
and 1000 iterations per scenario###
#------------------------------------------------------------------------------------------------#
for (m in 1:strat){
for (i in 1:iter){
for (t in 1:n.yrs){
  N[1,,i,m] <- N0                                                          #Initial abundances
    M.juv[t,,i,m] <- rmultinom(1,N[t,2,i,m],prob=c(p.M.juv,g.M.juv,(1-p.M.juv-g.M.juv)))
#Juveniles into p and g classes
  N[t+1,2,i,m] <- rbinom(1,N[t,1,i,m],s.M.hat) + M.juv[t,1,i,m]          #Juvs = new juvs + juvs
that survived and stayed
  N[t+1,3,i,m] <- rbinom(1,N[t,3,i,m],s.M.ad.yr[t,i,m]) + M.juv[t,2,i,m]                #Ads = new
ads + ads that survived and stayed
    F.juv[t,,i,m] <- rmultinom(1,N[t,5,i,m],prob=c(p.F.juv,g.F.juv,(1-p.F.juv-g.F.juv))) #Juveniles
into p and g classes
  N[t+1,5,i,m] <- rbinom(1,N[t,4,i,m],s.F.hat) + F.juv[t,1,i,m]                        #Juvs = new juvs +
juvs that survived and stayed
    N.F.cat[t,,i,m] <- rmultinom(1,N[t,6,i,m],prob=c(gam[t,i]*p.cross[t,i],gam[t,i]*(1-
p.cross[t,i]),1-gam[t,i])) #Ads into 3 classes: using JIC and cross, using JIC and nocross, and not
using JIC
    N.mgmt[t,i,m] <- rbinom(1,N.F.cat[t,1,i,m],p.mgmt[m])          #Crossers encounter mgmt area
    N.block[t,i,m] <- rbinom(1,N.mgmt[t,i,m],p.block[m])               #Crossers blocked by mgmt
    N.cross[t,i,m] <- N.F.cat[t,1,i,m] - N.block[t,i,m]                                  #Actual crossers
    N.nocross[t,i,m] <- N.F.cat[t,2,i,m] + N.block[t,i,m]                             #Actual no crossers
    N.surv.cross[t,i,m] <- rbinom(1,N.cross[t,i,m],s.F.cross[t,i,m])              #Crossers surviving
    N.surv.nocross[t,i,m] <- rbinom(1,N.nocross[t,i,m],s.F.nocross[t,i,m]   #Noncrossers surviving
    N.surv.nouser[t,i,m] <- rbinom(1,N.F.cat[t,3,i,m],s.F.nocross[t,i,m])        #Nonusers surviving
  N[t+1,6,i,m] <- N.surv.cross[t,i,m] + N.surv.nocross[t,i,m] + N.surv.nouser[t,i,m] +
F.juv[t,2,i,m]
    N.nester[t,i,m] <- N.surv.cross[t,i,m] + N.surv.nocross[t,i,m]              #Annual nesters on JIC
    N.nester.hab[t,,i,m] <- rmultinom(1,N.nester[t,i,m],prob=p.hab[m,])        #Nesters per habitat
    N.nest.hab[t,,i,m] <- N.nester.hab[t,,i,m]*clutch                            #Nests per habitat
    N.nest.surv.hab[t,,i,m] <- rbinom(3,N.nest.hab[t,,i,m],c(s.nest[m,1],s.nest[m,2],s.nest[m,3]))
#Surviving nests per habitat
    N.eggs.hab[t,,i,m] <- rpois(3,N.nest.surv.hab[t,,i,m]*clutch.size)                 #Eggs per habitat
    N.eggs.surv.hab[t,,i,m] <- rbinom(3,N.eggs.hab[t,,i,m],(s.egg*(s.M.hat^0.75)))       #Surviving
eggs per habitat
    N.F.new[t,,i,m] <-
rbinom(3,N.eggs.surv.hab[t,,i,m],c(fem.hatch.ratio[1],fem.hatch.ratio[2],fem.hatch.ratio[3]))
#New female hatchlings per habitat
    N.M.new[t,,i,m] <- N.eggs.surv.hab[t,,i,m] - N.F.new[t,,i,m] #New male hatchlings per habitat
```

```
  N[t+1,1,i,m] <- sum(N.M.new[t,,i,m])                              #New male hatchlings
  N[t+1,4,i,m] <- sum(N.F.new[t,,i,m])                              #New female hatchlings
  N[t+1,7,i,m] <- sum(N[t+1,1:6,i,m])                               #Annual total abundance
  N[t+1,7,i,m]<-(N[t,7,i,m]>49)*N[t+1,7,i,m]        #Makes N= 0 if falls below 50 (quasi-
extinction threshold)
  lambda[t,i,m] <- N[t+1,7,i,m]/N[t,7,i,m]          #Annual population growth rate
  lambda[lambda[,i,m]==0,i,m]<-NA                   #Replaces any 0s with NAs
  crash[i,m] <- min(which(is.na(lambda[,i,m])))         #Function to get time of crash (lambda=0
or NA) in each simulation
  crash[crash==Inf] <- 51
} #t
mean.lambda.iter[i,m]<- ifelse(crash[i,m]>20,mean(lambda[21:(crash[i,m]-1),i,m]),next)  #Mean
population growth rate per iteration
perc.change.iter[i,m] <- N[51,7,i,m]/N[1,7,i,m]         #Percentage change in abundance from
beginning of study
persist[i,m]<-(N[51,7,i,m]>0)*1                  #Indicator (0 or 1) if population persisted at year 50
persistence[i,m]<-mean(persist[i,m])
} #i
mean.lambda.mgmt[m] <- mean(mean.lambda.iter[,m],na.rm=T)       #Mean population growth
rate per strategy
perc.change.mgmt[m] <- mean(perc.change.iter[,m])      #Mean percentage change in abundance
from beginning of study
persistence.prob[m] <- mean(persist[,m],na.rm=T)    #Mean persistence probability per strategy
} #m
###End PVA Model###

persist[is.na(persist)] <- 0                  #May need if NAs appear in persist
#Results - main PVA
Lambda.PVA <- matrix(mean.lambda.mgmt,21,1)
Persist.PVA <- matrix(persistence.prob,21,1)

########################End of Code########################
```

APPENDIX D

DECISION FRAMEWORK DETAILS FOR MANAGEMENT ACTIONS AND CONSEQUENCES

LIST OF MANAGEMENT OPTIONS INCLUDED IN DECISION FRAMEWORK

The team of stakeholders involved in the structured decision making process for the Jekyll Island Causeway brainstormed management alternatives that would meet their objectives at an all-stakeholder workshop conducted on Jekyll Island in 2014. We grouped management options into eight categories that were expected to influence different objective(s) in the framework.

Table D1. List of management actions across eight categories included in the structured decision making framework for management of the Jekyll Island Causeway, Jekyll Island, GA. Actions are generally listed within a category from no or currently-deployed options to more intensive options. Note: all management options have some direct cost to JIA and their partners, and we specify the effects of actions on transferrable knowledge objectives in Table D3d.

Management categories	Actions	Description	Objective(s) impacted
Traffic	Current speed (55mph)		Road efficiency
	Inc. enforcement	No change to speed limit but increased enforcement on the JIC	
	Reduced speed (45mph)	Lower speed limit by 10 miles per hour	
Signage	Current (flashing warning signs)	Two flashing signs were permanently placed at opposite sides of the JIC in 2013, and they are programmed to flash during daily peaks of nesting activity around the diurnal high tide during nesting seasons	Risk awareness, conservation awareness, road aesthetics
	Speed displays	Digital signs that display driver's current speed	
	Scoreboards	LED scoreboards, modeled after signs in the GSTC, placed on the JIC to show drivers the current count of terrapins struck and saved that year	
Barriers	Hybrid barrier	A 22-m barrier composed of short fences and nest boxes was placed at 1 hot spot of terrapin crossing in 2011. Nest boxes are caged structures placed on raised mounds attractive to nesting females, and we electrified boxes to prevent raccoons from depredating nests	Number of terrapins on road, nest success, safety (bike path only), conservation awareness, road aesthetics
	Aversive (hot metal) strips	Experimental and untested barrier where flat metallic strips would be anchored on roadsides to absorb heat and create a thermal barrier that terrapins would not want to cross	
	Short fences (at hot spots or JIC-wide)	Corrugated tube-style fence < 0.5-m in height, placed ~5-10 m from the road surface that would allow drivers to pull off of the JIC if needed without running over fences	
	Terrapin gardens (cleared areas with fences and nest boxes at hot spots)	Hybrid strategy consisting of installing short fences and nest boxes in areas with cleared hedges	
	Bike path (north side or north & south side)	Constructed along the entire length of the JIC that includes a curb-style barrier along the marshside/grass side edge of the bike path	

Habitat	Current roadside hedge clearing	Current annual clearing of roadside hedges (42.4% of JIC roadsides clear) to maintain marsh vistas, conducted by the Jekyll Island Authority	Nest success, proportion of female hatchlings, conservation awareness, road aesthetics
	Moderate clearing	Increased levels of vegetation management (53.6% of JIC roadsides clear)	
	Intensive clearing	High levels of vegetation management (78.6% of JIC roadsides clear)	
Predators	None		Nest success, conservation awareness
	Aversive conditioning	Taste aversion strategies where some terrapin eggs are placed on the JIC prior to the nesting season and laced with a nonlethal chemical so that raccoons learn to associate all terrapin eggs with poor outcomes (illness)	
	Localized removal (hot spots)	Culling of raccoons using have-a-heart traps and lethal measures only at hot spots	
	JIC-wide removal	Using lethal measures on the JIC prior to and throughout the terrapin nesting season	
Education	Current (GSTC activities)	Includes daily outreach and education programs at the GSTC, distribution of terrapin bumper stickers, and occasional informational talks to local public groups	Risk awareness, conservation awareness, road aesthetics (interpretive signage only)
	Social media	Large-scale messaging of terrapin threats and management efforts using social media and local media coverage in newspapers and television	
	JI programs	Campaigns conducted by JIA/GSTC would target residents and employees through workshops to discuss terrapin conservation efforts and promote buy-in	
	Interpretive signage on JIC	Placed at frequently-used visitor pull-offs on the JIC to convey messages about terrapin ecology, threats, and management efforts	
	Radio programs	Radio service bulletin (similar to national parks) to transmit conservation messages to JIC drivers through a localized radio station	
Research	Current (GSTC activities)	Includes daily monitoring of nesting activity using road surveys and nest box inspection, rehabbing of injured females, recovering and incubating eggs, and headstarting and releasing of hatchlings at the GSTC	Targeted research
	Effects of fencing	Monitoring of terrapin nesting behavior, nesting success, and adult mortality in response to different barrier management options	
	Effects of predators	Monitoring of predator (especially raccoon) abundance, movement, and behavior related to management options.	

Financial	None		Conservation awareness, external funds
	Grants	Collaborative grant writing between UGA and GSTC researchers for federal and state agencies and special interest groups	
	Fundraisers	Conduct fundraiser event on Jekyll Island in collaboration with JIA, GSTC, and UGA to gain local and corporate donor support	
	Conservation fees for road users	Establish small conservation fee (~$1) per vehicle entering Jekyll Island	

COMPLETED CONDITIONAL PROBABILITY TABLES AND DIRECTLY ESTIMATED CONSEQUENCES FOR BAYESIAN BELIEF NETWORK

Subsets of scientific researchers, professionals in state agencies, local managers, and public representatives were solicited to estimate the outcomes of actions or objectives related to management of the Jekyll Island Causeway, Jekyll Island, GA that fell within their areas of expertise. Experts provided two types of estimates that were used to parameterize the Bayesian Belief Network: 1) conditional probability tables regarding outcomes of actions or objectives (Table D2a-c), and 2) direct estimates of consequences of single actions or combinations of actions included in the list of management strategies (Table D3a-e). We report the actions specified in each management strategy (Table D4a) and estimated cumulative effects of each strategy on objectives (Table D4b-d). For conditional probability tables, we report mean and group-specific estimates, and we averaged group-specific values when more than one expert from a group provided estimates. For direct estimates, we report mean “best estimates” and standard deviations of each estimate. We also report the effect size of each action, indicating the difference between outcomes of that action with baseline outcomes (first row in each table). We used means and standard deviations to calculate a probability distribution for the decision network via the method of moments. To limit the response burden, we often asked for estimates for a range of actions or objective states – but not every action, combination, or objective state. We used the range of estimates to interpolate conditional probabilities and action consequences for cases not directly estimated by experts, and we denote these values in all tables with an asterisk (*).

Table D2a. Conditional probabilities for attributes of **safety** (relative number of accidents on the causeway), given states of influencing objectives (level of vehicle control, number of interveners on the road, and presence of a bike path on the north or north and south sides of the road.

Vehicle control	**No. of Interv.**	**Bike path**	Group mean (N = 3)			GDOT (n = 1)			JI managers (n = 2)		
			Fewer	**Current**	**More**	**Fewer**	**Current**	**More**	**Fewer**	**Current**	**More**
Low	0-50	None	72.5	27.5	0	70	30	0	75	25	0
Low	0-50	North*	81.5	13.5	5	75	15	10	88	12	0
Low	0-50	N&S*	90	6.25	3.75	85	7.5	7.5	95	5	0
Mod	0-50	None	86	14	0	82	18	0	90	10	0
Mod	0-50	North*	92.5	5	2.5	90	5	5	95	5	0
Mod	0-50	N&S*	95	3.125	1.875	90	6.25	3.75	100	0	0
High	0-50	None	92.5	7.5	0	90	10	0	95	5	0
High	0-50	North*	97	3	0	94	6	0	100	0	0
High	0-50	N&S*	98.5	1.5	0	97	3	0	100	0	0
Low	50-100	None	45	55	0	50	50	0	40	60	0
Low	50-100	North*	70	25	5	60	30	10	80	20	0
Low	50-100	N&S*	87.5	8.75	3.75	75	17.5	7.5	100	0	0
Mod	50-100	None	65	35	0	50	50	0	80	20	0
Mod	50-100	North*	82.5	15	2.5	70	25	5	95	5	0
Mod	50-100	N&S*	92.5	5.625	1.875	85	11.25	3.75	100	0	0
High	50-100	None	80	20	0	70	30	0	90	10	0
High	50-100	North*	90	10	0	85	15	0	95	5	0
High	50-100	N&S*	95	5	0	90	10	0	100	0	0
Low	100+	None	0	100	0	0	100	0	0	100	0
Low	100+	North	50	45	5	40	50	10	60	40	0
Low	100+	N&S	77.5	18.75	3.75	55	37.5	7.5	100	0	0
Mod	100+	None	20	80	0	0	100	0	40	60	0
Mod	100+	North*	62.5	35	2.5	45	50	5	80	20	0
Mod	100+	N&S*	82.5	15.625	1.875	65	31.25	3.75	100	0	0
High	100+	None	35	65	0	0	100	0	70	30	0
High	100+	North*	70	30	0	50	50	0	90	10	0
High	100+	N&S*	85	15	0	70	30	0	100	0	0

Table D2b. Conditional probabilities for the **number interveners**, given states of influencing objectives (number of terrapins on the road).

	Group mean (N = 6)			GDOT (n = 1)			JI managers (n = 3)			JI public (n = 2)		
No. terps on road	**0-50**	**50-100**	**100+**	**0-50**	**50-100**	**100+**	**0-50**	**50-100**	**100+**	**0-50**	**50-100**	**100+**
0-50	85.5	8.8	5.7	33	33	34	93.3	6.7	0	100	0	0
50-100	44.2	29.2	26.7	15	10	75	41.7	36.7	21.7	62.5	27.5	10
100-200	9.2	40	50.8	5	10	85	10	33.3	56.7	10	65	25
200+	0.8	6.7	92.5	0	5	95	1.7	8.3	90	0	5	95

Table D2c. Conditional probabilities for levels of **vehicle control**, given states of influencing objectives (number of terrapins on the road, percentage of drivers aware of risks, and road efficiency [speed])

No. terps	**Risk awareness**	**Road efficiency**	Group mean (N = 3)			GDOT (n = 1)			JI managers (n = 2)		
			Low	**Mod**	**High**	**Low**	**Mod**	**High**	**Low**	**Mod**	**High**
0-50	0-20	High	12.5	17.5	70	12.5	17.5	70	12.5	17.5	70
0-50	0-20	Mod*	12.3	17.5	70.2	11.9	17.6	70.5	12.5	17.5	70
0-50	0-20	Low*	10.4	17.9	71.7	8.8	18.3	73	11.3	17.8	71
0-50	20-40	High*	10.8	17.8	71.3	11.3	17.8	71	10.6	17.9	71.5
0-50	20-40	Mod*	10.5	17.9	71.6	9.7	18.1	72.3	10.9	17.8	71.3
0-50	20-40	Low*	9.6	18.1	72.3	10	18	72	9.4	18.1	72.5
0-50	40-60	High*	9.2	18.2	72.7	10	18	72	8.8	18.3	73
0-50	40-60	Mod*	8.8	18.3	73	7.5	18.5	74	9.4	18.1	72.5
0-50	40-60	Low*	8.8	18.3	73	11.3	17.8	71	7.5	18.5	74
0-50	60-80	High*	8.1	18.4	73.5	9.4	18.1	72.5	7.5	18.5	74
0-50	60-80	Mod*	7.7	18.5	73.8	6.9	18.6	74.5	8.1	18.4	73.5
0-50	60-80	Low*	7.1	18.6	74.3	8.8	18.3	73	6.3	18.8	75
0-50	80-100	High*	7.1	18.6	74.3	8.8	18.3	73	6.3	18.8	75
0-50	80-100	Mod*	6.7	18.7	74.7	6.3	18.8	75	6.9	18.6	74.5
0-50	80-100	Low*	5.4	18.9	75.7	6.3	18.8	75	5	19	76
50-100	0-20	High	37.5	20.8	41.7	37.5	20.8	41.7	37.5	20.8	41.7
50-100	0-20	Mod*	36.9	21.0	42.1	35.6	21.5	42.9	37.5	20.8	41.7
50-100	0-20	Low*	31.3	22.9	45.8	26.3	24.6	49.2	33.8	22.1	44.2
50-100	20-40	High*	32.5	22.5	45	33.8	22.1	44.2	31.9	22.7	45.4
50-100	20-40	Mod*	31.6	22.8	45.6	29.1	23.6	47.3	32.8	22.4	44.8
50-100	20-40	Low*	28.8	23.8	47.5	30	23.3	46.7	28.1	24.0	47.9
50-100	40-60	High*	27.5	24.2	48.3	30	23.3	46.7	26.3	24.6	49.2
50-100	40-60	Mod*	26.3	24.6	49.2	22.5	25.8	51.7	28.1	24.0	47.9
50-100	40-60	Low*	26.3	24.6	49.2	33.8	22.1	44.2	22.5	25.8	51.7
50-100	60-80	High*	24.4	25.2	50.4	28.1	24.0	47.9	22.5	25.8	51.7
50-100	60-80	Mod*	23.1	25.6	51.3	20.6	26.5	52.9	24.4	25.2	50.4
50-100	60-80	Low*	21.3	26.3	52.5	26.3	24.6	49.2	18.8	27.1	54.2
50-100	80-100	High*	21.3	26.3	52.5	26.3	24.6	49.2	18.8	27.1	54.2
50-100	80-100	Mod*	20	26.7	53.3	18.8	27.1	54.2	20.6	26.5	52.9
50-100	80-100	Low*	16.3	27.9	55.8	18.8	27.1	54.2	15	28.3	56.7
100-200	0-20	High	66.6	22.3	11.1	66.6	22.3	11.1	66.6	22.3	11.1
100-200	0-20	Mod*	65.5	23.0	11.5	63.3	24.5	12.2	66.6	22.3	11.1
100-200	0-20	Low*	55.5	29.7	14.8	46.6	35.6	17.8	59.9	26.7	13.4
100-200	20-40	High*	57.7	28.2	14.1	59.9	26.7	13.4	56.6	28.9	14.5
100-200	20-40	Mod*	56.1	29.3	14.6	51.6	32.3	16.1	58.3	27.8	13.9
100-200	20-40	Low*	51.1	32.6	16.3	53.3	31.1	15.6	50.0	33.4	16.7
100-200	40-60	High*	48.8	34.1	17.1	53.3	31.1	15.6	46.6	35.6	17.8
100-200	40-60	Mod*	46.6	35.6	17.8	40.0	40.0	20.0	50.0	33.4	16.7
100-200	40-60	Low*	46.6	35.6	17.8	59.9	26.7	13.4	40.0	40.0	20.0
100-200	60-80	High*	43.3	37.8	18.9	50.0	33.4	16.7	40.0	40.0	20.0

100-200	60-80	Mod*	41.1	39.3	19.6	36.6	42.2	21.1	43.3	37.8	18.9
100-200	60-80	Low*	37.7	41.5	20.8	46.6	35.6	17.8	33.3	44.5	22.2
100-200	80-100	High*	37.7	41.5	20.8	46.6	35.6	17.8	33.3	44.5	22.2
100-200	80-100	Mod*	35.5	43.0	21.5	33.3	44.5	22.2	36.6	42.2	21.1
100-200	80-100	Low*	28.9	47.4	23.7	33.3	44.5	22.2	26.6	48.9	24.5
200+	0-20	High	100	0	0	100	0	0	100	0	0
200+	0-20	Mod	98.3	1.7	0	95	5	0	100	0	0
200+	0-20	Low	83.3	13.3	3.3	70	20	10	90	10	0
200+	20-40	High*	86.7	9.2	4.2	90	7.5	2.5	85	10	5
200+	20-40	Mod*	84.2	10.8	5	77.5	12.5	10	87.5	10	2.5
200+	20-40	Low*	76.7	16.7	6.7	80	15	5	75	20	5
200+	40-60	High	73.3	18.3	8.3	80	15	5	70	20	10
200+	40-60	Mod	70	20	10	60	20	20	75	20	5
200+	40-60	Low	70	20	10	90	10	0	60	30	10
200+	60-80	High*	65	24.2	10.8	75	17.5	7.5	60	27.5	12.5
200+	60-80	Mod*	61.7	25.8	12.5	55	22.5	22.5	65	27.5	7.5
200+	60-80	Low*	56.7	29.2	14.2	70	17.5	12.5	50	37.5	12.5
200+	80-100	High	56.7	30	13.3	70	20	10	50	35	15
200+	80-100	Mod	53.3	31.7	15	50	25	25	55	35	10
200+	80-100	Low	43.3	38.3	18.3	50	25	25	40	45	15

Table D3a. Consequences of actions on **percentage of drivers aware of risks** on the Jekyll Island Causeway, Jekyll Island, GA. Respondents (N = 7) estimated percentage of drivers aware, and we used the method of moments to convert the mean and variance into a probability distribution.

Action categories	Actions	Mean	St. dev.	Effect size
Signage	Current (static and flashing yellow signs)	50.0	21.8	-
	Current, speed displays	64.3	21.7	14.3
	Current, scoreboards (# of terrapins hit vs. saved)	69.3	20.7	19.3
	All signs	76.4	23.8	26.4
Education	Interpretive signage*	55.0	19.5	5.0
	Targeted JI programs	59.3	21.3	9.3
	Local and social media messages*	55.7	19.5	5.7
	Radio programs*	55.0	22.0	5.0
	Local social media, targeted JI programs	65.0	21.2	15.0
	Local social media, interpretive signage	57.1	19.5	7.1
	Local social media, targeted JI programs, interpretive signage, radio programs*	67.9	22.0	17.9
Signage and education				
Current + "Your speed"	Local social media, targeted JI programs	72.1	20.8	22.1
	Local social media, interpretive signage	70.0	18.5	20.0
Current + Scoreboards	Local social media, targeted JI programs	75.7	19.5	25.7
	Local social media, targeted JI programs, interpretive signage, radio programs	80.0	22.0	30.0
All road signs	Local social media, targeted JI programs, interpretive signage	80.7	20.9	30.7
	Local social media, targeted JI programs, radio programs	80.7	21.3	30.7

Table D3b. Consequences of actions on percentage of **patrons aware and supportive of conservation efforts**. Respondents (N = 17) estimated percentage of patrons aware and supportive, given actions, We used the method of moments to convert the mean and variance into a probability distribution.

Action categories	Actions and combinations	Mean	St. dev.	Effect size
Signage	Current (static and flashing yellow signs)	54.4	15.3	-
	Current, scoreboards	69.0	14.3	14.6
Predators	Local aversive	46.7	14.7	-7.7
	Localized removal	35.0	11.8	-19.4
	JIC-wide removal	31.7	12.5	-22.7
Education	Interpretive signage*	57.7	13.3	3.3
	Targeted JI programs	63.5	13.3	9.1
	Local and social media messages*	56.3	13.3	1.9
	Radio programs*	59.6	15.7	5.2
	Local social media, targeted JI programs	65.4	15.0	11.0
	Interpretive signage, social media	64.8	15.7	10.4
	Interpretive signage, targeted campaigns, social media, radio programs*	72.5	15.7	18.1
Financial	Conservation fees	28.3	20.5	-17.2
	Fundraisers	54.2	21.2	15.6
	Conservation fees and fundraisers	45.8	22.2	-8.6
Signage and education				
Current signage + Scoreboards (# of terrapins struck vs. saved)	Local social media, targeted JI programs	71.6	16.9	17.2
	Local social media, targeted JI programs, interpretive signage	74.9	17.0	20.5
	Local social media, targeted JI programs, radio programs	77.5	11.9	23.1
	Local social media, targeted JI programs, interpretive signage, radio programs	80.1	12.5	25.7
Predator management and education				
Aversive conditioning	Current education	46.7	14.7	-7.7
	Targeted JI programs	54.2	15.6	-0.2
	Local social media, targeted JI programs	61.7	13.7	7.3
	Above, plus interpretive signage	71.7	11.3	17.3
Local lethal removal	Current education	35.0	11.8	-19.4
	Targeted JI programs	38.3	9.3	-16.1
	Local social media, targeted JI programs	42.5	8.2	-11.9
	Above, plus interpretive signage	48.3	6.8	-6.1
JIC-wide lethal removal	Current education	31.7	12.5	-22.7
	Targeted JI programs	35.8	9.2	-18.6
	Local social media, targeted JI programs	40.8	8.0	-13.6
	Above, plus interpretive signage	44.2	9.2	-10.2

Table D3c. Consequences of actions on **road aesthetics**. Respondents (N = 6) rated the aesthetic outcomes of visible management actions. Constructed attribute scale: 1 = worst, 2 = worse, 3 = current, satisfactory, 4 = better, and 5 = best aesthetic conditions. We used the method of moments to convert the mean and variance into a probability distribution.

Action categories	**Actions and combinations**	**Mean**	**St. dev.**	**Effect size**
Signage	Current signage	3.0	0	-
	Current + "Your speed"	3.3	0.52	0.3
	Current + Scoreboards	3.2	0.75	0.2
	All road signs	3.2	1.17	0.2
Barriers	Hybrid barrier (nest boxes and short fences) at 1 hot spot with few additional nest boxes	3.0	0	0.0
	Aversive (hot metal) strips at 3 hot spots	2.7	0.52	-0.3
	Short fences (hot spots)	2.5	0.55	-0.5
	Terrapin gardens (hot spots)	3.2	0.41	0.2
	Terrapin gardens with nest boxes (hot spots)	3.7	1.03	0.7
	Short fences N side (hot spots), terrapin gardens S side (hot spots)	3.0	0	0.0
	Bike path (entire north side of JIC)	3.3	0.82	0.3
	Bike path N side, terrapin gardens (hot spots) S side	3.3	0.52	0.3
	Bike path N side, terrapin gardens with nest boxes (hot spots) S side	3.3	0.52	0.3
	Bike path N side, fences hot and warm spots S side	2.5	0.55	-0.5
	Bike path N side, terrapin gardens (hot spots) and fences (warm spots) S side	2.5	0.55	-0.5
	Bike path N side, fences (JIC wide) S side	2.0	0.63	-1.0
	Bike path (entire north and south side of JIC)	3.5	1.05	0.5
	Bike path N&S side, nest boxes at 3 hotspots N&S side	3.7	0.82	0.7
Vegetation	Current vista clearing regime (42% of roadsides clear)	3.0	0	0.0
	Moderate vista clearing (54% clear)	3.7	0.82	0.7
	High vista clearing (79%)	2.5	0.55	-0.5
Education	Interpretive signage on at towers	3.8	0.75	0.8

Table D3d. Consequences of actions on means objectives influencing **transferable knowledge gained**. Constructed attribute scale: 0 = lower, 1 = current, 2 = higher, and 3 = highest degree of knowledge gained from management and research.

Action categories	Actions	Targeted research	Mgmt for terps	Mgmt for species
Traffic	Current (55mph)	1	1	1
	Enforcement	1	1	1
	45mph	1	1	1
Signage	Current (static and flashing yellow signs)	1	1	1
	Current, speed displays	1	1	2
	Current, scoreboards	1	1	2
Barriers	Current (hybrid barrier with short fences and nest boxes at 1 hot spots)	1	1	1
	Short fences (hot spots, 10% of JIC)	1	2	1
	Short fences (hot and warm spots, 30% of JIC)	1	2	1
	Short fences (entire JIC)	1	1	2
	Nest boxes (hot spots)	1	2	1
	Terrapin gardens (HS)	1	2	1
	Terrapin gardens (HS) and short nest boxes	1	3	1
	Aversive metal strips	1	1	3
	Bike Path (N side)	1	1	2
	Bike Path (N&S side)	1	1	2
Habitat	Current clearings + annual rotation of 2 linear km (CC 2km) - 42% of roadside clear	1	1	1
	Current clearings + 4km - 54% of roadside clear	1	1	1
	Current clearings + 8.5km - 79% of roadside clear	1	1	1
Predators	None	1	1	1
	Local aversive	1	1	2
	Localized removal	1	1	2
	JIC-wide removal	1	1	2
Education	Current	1	1	1
	Interpretive signage	1	1	1
	Targeted JI programs	1	1	1
	Local and social media messages	1	1	1
	Radio programs	1	1	1
	Ed against intervening	1	1	1
Research	Current	1	1	1
	Reduced effort	0	1	1
	More projects on effects of fencing	2	1	1
	More projects on effects of predators	2	1	1
Financial	None	1	1	1
	Conservation fees	1	1	1
	Grants	1	1	1
	Fundraisers	1	1	1

Table D3e. Estimated direct costs and external funds for actions related to management of the Jekyll Island Causeway, Jekyll Island, GA. Costs were estimated using online resources and personal communications with local researchers and managers.

Action categories	Actions	Direct Costs	External funds	Description	Source
Traffic	Current (55mph)	$ -		-	-
	Enforcement	$ -		Reallocation of effort of GA State Patrol, costs not incurred by project	Ben Carswell (pers. comm)
	45mph	$ -		Costs not incurred by project, requires assessment from GDOT	GDOT (pers. comm)
Signage	Current (static and flashing yellow signs)	$ 5,350		2 flashing signs and equipment.	Carmanah Solutions, http://carmanah.com/traffic/solar-flashing-beacons
	Speed displays	$ 7,032		Price reflects 2 sign kits (sign with LED display, solar panel, battery console, software)	Tapco, http://www.globalindustrial.com/p/safety/signs/parking-traffic/15-character-7-segment-blinkerradar-led-feedback-sign-white-110v-ac
	Scoreboards (# terrapins dead vs. saved)	$ 10,050		Price reflects 2 sign kits (sign with LED display, solar panel, battery console, software)	All Traffic Solutions
Barriers	Current	$ 2,597		Hybrid barrier with short fences and nest boxes at 1 hot spot. Materials for 6 nest boxes, Tenax fencing, wooden stakes, soil for mound construction, vegetation, electric system, and labor for installation and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)
	Short fences (hot spots, 10% of JIC)	$ 4,862		Tenax fencing, wooden stakes, and labor for installation and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)
	Short fences (hot and warm	$ 13,456		Tenax fencing, wooden stakes, and labor for installation and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)

spots, 30% of JIC)			
Short fences (entire JIC)	$ 55,414	Tenax fencing, wooden stakes, and labor for installation and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)
Nest boxes (hot spots)	$ 5,195	Materials for 12 additional nest boxes, Tenax fencing, soil for mound construction, vegetation, electric system, and labor for installation and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)
Terrapin gardens (HS)	$ 4,861	Tenax fencing, wooden stakes, vegetation, and labor for hedge clearing, installation, and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)
Terrapin gardens (HS) and short nest boxes	$ 9,515	Materials for 12 additional short nest boxes, Tenax fencing, wooden stakes, vegetation, electric system, and labor for hedge clearing, installation, and maintenance.	Cliff Gawron, Kurt Buhlmann (pers. comm)
Aversive metal strips	$ 5,205	Materials for placing metal flashing along roadsides at 3 hotspots	http://www.homedepot.com/p/Gibraltar-Building-Products-3-in-x-3-in-x-10-ft-Galvanized-Steel-Roof-Edge-Flashing-15553/202092840?N=5yc1vZaqp7
Bike Path (N side)	$ 250,456	Reflects 20% contribution of total project costs from JIF (GDOT contributes 80%). Includes materials for construction and planning and permitting fees. Extrapolated from cost of $312,500 for construction of 1.37-mi bike path on north side of causeway between JI Bridge and Guest Information Center.	Ben Carswell (pers. comm)
Bike Path (N&S side)	$ 500,912	Reflects 20% contribution of total project costs from JIF (GDOT contributes 80%). Includes materials for construction and planning and permitting fees.	Ben Carswell (pers. comm)

			Extrapolated from cost of $312,500 for construction of 1.37-mi bike path on north side of causeway between JI Bridge and Guest Information Center.	
Habitat	Current clearing (42% of roadside)	$ 30,000	Annual rotation of 2 linear km in addition to permanently cleared areas. Includes labor and fuel costs to annually clear roadside hedges by JIA.	Cliff Gawron (pers. comm)
	Moderate clearings (54%)	$ 58,679	Annual rotation of 4 linear km in addition to permanently cleared areas. Includes labor and fuel costs to annually clear roadside hedges by JIA.	Cliff Gawron (pers. comm)
	Significant clearings (79%)	$ 121,771	Annual rotation of 8.5 linear km in addition to permanently cleared areas. Includes labor and fuel costs to annually clear roadside hedges by JIA.	Cliff Gawron (pers. comm)
Predators	None	$ -	-	-
	Local aversive	$ 1,263	20mg of emetine dihydrochloride per laced egg, labor and material costs for distributing 36 eggs across hotspots each week for 6 weeks preceding and during beginning of each nesting season	Sigma Aldrich, http://www.sigmaaldrich.com/catalog/product/sigma/e2375?lang=en&region=US
	Localized removal	$ 1,394	Labor for two in-house employees	Ben Carswell (pers. comm)
	JIC-wide removal	$ 14,000	Includes equipment and labor for consulting agency	Environmental 360 (pers. comm)
Education	Current	$ 50,800	Includes in-house programs, facebook, bumper stickers, etc.	Katie Higgins, Lori Hunt (pers. comm)
	Interpretive signage	$ 2,000	Includes material costs for panel and frame, labor costs for installation	Ben Carswell (pers. comm)
	Targeted JI programs	$ 530	Includes printing costs and labor for 1 staff and 1 Americorps member to develop and complete 5 1-hr programs per summer	Greg Skupien (pers. comm), Vista Print

	Local and social media messages	$ 3,477		Includes labor and material costs for increased social media (2 hrs/wk during summer) and increased development of special projects (writing 4 news releases or local media articles per summer, working with public broadcasting or other TV outlets, and creating original video content, similar to crowdfunding, for online distribution)	Katie Higgins, Lori Hunt, David Zailo (pers. comm)
	Radio programs	$ 24,744		Includes full installation and activation of Traveler's Information Station	Information Station Specialists
	Ed against intervening	$ -		Only requires modifying messages of current education programs	
Research	Current	$ 94,801		Full-time salary for 2 graduate students and 1 Americorps Member, additional labor from GSTC staff, fuel and mileage costs for road surveying, misc. materials	Michelle Kaylor, Kimberly Andrews (pers. comm)
	Reduced effort	$ 65,883		Assumes reduced causeway patrols, maintenance of signs and nest boxes, and time of GSTC staff and AC member dedicated to field terrapin research	Michelle Kaylor, Kimberly Andrews (pers. comm)
	More projects on effects of fencing	$ 19,800		Half-time salary for 1 additional graduate student and 1 additional Americorps Member	Michelle Kaylor, Kimberly Andrews (pers. comm)
	More projects on effects of predators	$ 19,800		Half-time salary for 1 additional graduate student and 1 additional Americorps Member	Michelle Kaylor, Kimberly Andrews (pers. comm)
Financial	None		-	-	-
	Conservation fees		$ 38,199	Includes $5 increase to annual pass and $1 increase to daily passes to Jekyll Island, based on passes sold in 2012	JIA (unpubl. data)
	Grants		$ 17,936	Based on writing 4 grants per year that covers range of monetary funds (from $2,500 to $400,000). Estimate	John Maerz (pers. comm)

		incorporates labor costs of PI for writing time and chance of receiving each grant.	
Fundraisers	$ 17,500	Assumes gain of $25,000 and cost of $7,500 for large fundraiser held on Jekyll Island or St. Simons	John Maerz (pers. comm)

Table D4a. List of management strategies for the Jekyll Island Causeway, Jekyll Island, GA created through all-stakeholder workshops.

Strategy (type)	Targeted Objectives (or themes)	Action categories							
		Traffic	**Signage**	**Barriers**	**Habitat**	**Predators**	**Education**	**Research**	**Financial**
1 (Status quo)	Maintain current mgmt., ed., & research actions	55mph	Static, Flashing	Hybrid barrier (short fences and nest boxes) at 1 hot spot	Current % of roadside undergoing vista clearing (VC 42%)	None	Current	Current	None
2 (Reduced effort)	Decreased maintenance & research effort	55mph	Static, Flashing	Hybrid barrier (short fences and nest boxes) at 1 hot spot	VC 42%	None	Current	Reduced effort	None
3 (Tradeoff)	Terrapin persistence	55mph	Current	Bike path N&S side, nest boxes at 3 hotspots N&S	VC 79%	JIC-wide removal	Current	Current	None
4 (Tradeoff)	Terrapin - road mortality	55mph	Current	Bike path N&S	VC 42%	None	Current	Barriers	None
5 (Tradeoff)	Terrapin - nest predation	55mph	Current	Terrapin gardens with nest boxes (hot spots) S side	VC 79%	JIC-wide removal, local aversive	Current	Predators	None
6 (Tradeoff)	Safety	45mph	All signs	Bike path N side, fences (JIC wide) S side	VC 42%	None	Radio, ed against intervening	Current	None
7 (Tradeoff)	Satisfaction	55mph	Current, scoreboard	Current	VC 42%	None	All but against intervening	Current	None
8 (Tradeoff)	Info Gained	55mph	All signs	Aversive strips	VC 79%	JIC-wide removal, local aversive	Current	All	None
9 (Tradeoff)	Cost	55mph	Current	Current	VC 42%	None	Current	Current	All
10 (Compromise)	Moderate actions on all objectives	Enforcement	Current, speed display	Short fences N side (hot spots), terrapin gardens	VC 54%	Localized removal	Interpretive signage, social media	All	Cons fees, Grants

				S side (hot spots)					
11 (Compromise)	Low cost, localized terp mgmt	Enforce-ment	Current, speed display	Fences only (hotspots)	VC 54%	Local aversive	Local social media, targeted JI programs	All	Grants
12 (Compromise)	Moderate safety & satisfaction	Enforce-ment	Current, scoreboard	Terrapin gardens (hot spots)	VC 42%	None	Ed against intervening	Current	None
13 (Hybrid)	Short barriers, local mgmt	55mph	Current	Bike path N side	VC 54%	Localized removal	Targeted JI programs	All	Grants
14 (Hybrid)	Short barriers, broader mgmt, education, awareness	55mph	All signs	Bike path N side	VC 42%	JIC-wide removal	Interpretive signage, ed against intervening	All	Cons fees, Grants
15 (Hybrid)	Medium barriers, local mgmt	55mph	Current	Bike path N side, terrapin gardens (hot spots) S side	VC 42%	None	Targeted JI programs	All	Grants, fundraisers
16 (Hybrid)	Medium barriers, medium mgmt, education, awareness	55mph	Current, scoreboard	Bike path N side, terrapin gardens with nest boxes (hot spots) S side	VC 54%	Localized removal	Local social media, targeted JI programs	All	Cons fees, Grants
17 (Hybrid)	Medium barriers, broader mgmt, education, awareness	55mph	All signs	Bike path N side, terrapin gardens (hot spots) S side	VC 54%	JIC-wide removal	Interpretive signage, ed against intervening	All	Cons fees, Grants
18 (Hybrid)	Longer barriers, local mgmt	55mph	Current	Bike path N side, fences hot and warm spots S side	VC 42%	None	Targeted JI programs	All	Grants, fundraisers
19 (Hybrid)	Longer barriers, medium mgmt, education	55mph	Current	Bike path N side, terrapin gardens (hot spots) and fences (warm spots) S side	VC 54%	Localized removal	Local social media, targeted JI programs	All	Grants, fundraisers

20 (Hybrid)	Longer barriers, broader mgmt, education	55mph	Current	Bike path N side, fences (JIC-wide) S side	VC 54%	JIC-wide removal	Interpretive signage, local social media	All	Grants, fundraisers

Table D4b. Predicted consequences of management strategies on components influencing the fundamental objective of maximizing terrapin population persistence. Note: survival estimates of crossing females were influenced by the number of terrapins emerging on the road and level of driver awareness.

	Nest success			**No. terrapins on road**			**Increase in survival of crossing females**		
Strategy (type)	LCI	Mean	UCI	LCI	Mean	UCI	LCI	Mean	UCI
1 (Status quo)	0.430	0.530	0.620	282	286	293	-	-	-
2 (Reduced effort)	0.430	0.530	0.620	282	286	293	-	-	-
3 (Tradeoff)	0.762	0.856	0.927	18	58	102	-	-	-
4 (Tradeoff)	0.395	0.500	0.582	56	88	123	-	-	-
5 (Tradeoff)	0.762	0.856	0.927	180	210	235	-	-	-
6 (Tradeoff)	0.395	0.500	0.582	70	133	175	0.009	0.035	0.056
7 (Tradeoff)	0.430	0.530	0.620	282	286	293	0.005	0.025	0.042
8 (Tradeoff)	0.752	0.846	0.917	222	240	260	0.005	0.025	0.042
9 (Tradeoff)	0.430	0.530	0.620	282	286	293	-	-	-
10 (Compromise)	0.587	0.701	0.787	208	232	251	0.009	0.035	0.056
11 (Compromise)	0.451	0.553	0.631	215	243	261	0.009	0.035	0.056
12 (Compromise)	0.373	0.473	0.577	219	239	255	0.009	0.035	0.056
13 (Hybrid)	0.627	0.741	0.827	199	213	229	-	-	-
14 (Hybrid)	0.673	0.764	0.850	212	223	235	0.005	0.025	0.042
15 (Hybrid)	0.393	0.493	0.597	149	176	202	-	-	-
16 (Hybrid)	0.627	0.741	0.827	122	160	190	0.005	0.025	0.042
17 (Hybrid)	0.685	0.782	0.869	122	160	190	0.005	0.025	0.042
18 (Hybrid)	0.383	0.483	0.583	128	170	200	-	-	-
19 (Hybrid)	0.616	0.733	0.824	109	149	183	-	-	-
20 (Hybrid)	0.680	0.777	0.864	57	123	168	-	-	-

Table D4c. Predicted consequences of management strategies on components influencing the fundamental objectives of maximizing patron safety and satisfaction.

	Safety			**Satisfaction**				
	% of drivers aware of risks			**% of patrons aware and supportive of conservation efforts**			**Aesthetic rating (0 = worst, 0.5 = satisfactory, 1 = best)**	
Strategy	LCI	Mean	UCI	LCI	Mean	UCI	Mean	SD
1 (Status quo)	34.3	50.0	62.1	40.6	54.4	65.6	0.500	0.010
2 (Reduced effort)	34.3	50.0	62.1	29.6	43.4	54.6	0.500	0.010
3 (Tradeoff)	34.3	50.0	62.1	19.2	31.7	44.2	0.292	0.025
4 (Tradeoff)	34.3	50.0	62.1	40.6	54.4	65.6	0.500	0.018
5 (Tradeoff)	34.3	50.0	62.1	11.2	23.7	36.2	0.667	0.025
6 (Tradeoff)	60.0	76.4	85.7	60.1	75.0	85.7	0.583	0.025
7 (Tradeoff)	62.9	80.0	86.4	65.1	80.1	88.1	0.583	0.012
8 (Tradeoff)	60.0	76.4	85.7	34.2	46.7	59.2	0.708	0.025
9 (Tradeoff)	34.3	50.0	62.1	37.5	45.8	57.5	0.500	0.000
10 (Compromise)	56.4	70.0	78.6	16.7	28.3	42.5	0.565	0.021
11 (Compromise)	60.7	72.1	80.7	48.3	61.7	74.2	0.292	0.021
12 (Compromise)	53.6	69.3	80.0	34.2	46.7	59.2	0.500	0.013
13 (Hybrid)	43.6	59.3	70.7	25.0	38.3	50.0	0.625	0.021
14 (Hybrid)	65.0	80.7	87.1	21.2	33.7	46.2	0.958	0.018
15 (Hybrid)	43.6	59.3	70.7	68.3	85.8	94.2	0.583	0.017
16 (Hybrid)	61.4	75.7	82.9	21.7	35.6	43.8	0.500	0.017
17 (Hybrid)	65.0	80.7	87.1	21.2	33.7	46.2	0.940	0.021
18 (Hybrid)	43.6	59.3	70.7	68.3	85.8	94.2	0.500	0.020
19 (Hybrid)	52.1	65.0	76.4	46.8	58.5	71.8	0.625	0.022
20 (Hybrid)	42.9	57.1	66.4	39.5	51.2	65.3	0.667	0.021

Table D4d. Predicted consequences of management strategies on components influencing the fundamental objectives of maximizing transferrable knowledge and minimizing net cost.

	Transferrable knowledge				Net cost		
Strategy	Targeted research	Mgmt for terrapins	Mgmt for species	Total knowledge score	Direct cost	External support	Net cost
1 (Status quo)	1	1	1	3	$ 183,548	0	$ 183,548
2 (Reduced effort)	1	1	1	3	$ 154,631	0	$ 154,631
3 (Tradeoff)	1	3	3	7	$ 795,426	0	$ 795,426
4 (Tradeoff)	2	1	2	5	$ 704,260	0	$ 704,260
5 (Tradeoff)	2	3	3	8	$ 319,897	0	$ 319,897
6 (Tradeoff)	1	1	3	5	$ 531,244	0	$ 531,244
7 (Tradeoff)	1	1	2	4	$ 224,349	0	$ 224,349
8 (Tradeoff)	3	3	3	9	$ 349,871	0	$ 349,871
9 (Tradeoff)	1	1	1	3	$ 183,548	$73,635	$ 109,913
10 (Compromise)	3	2	3	8	$ 273,022	$56,135	$ 216,888
11 (Compromise)	3	2	3	8	$ 268,991	$17,936	$ 251,055
12 (Compromise)	1	2	2	5	$ 198,459	0	$ 198,459
13 (Hybrid)	3	1	3	7	$ 504,207	$17,936	$ 486,271
14 (Hybrid)	3	1	3	7	$ 506,686	$56,135	$ 450,551
15 (Hybrid)	3	2	2	7	$ 478,995	$35,436	$ 443,559
16 (Hybrid)	3	3	3	9	$ 527,249	$56,135	$ 471,114
17 (Hybrid)	3	2	3	8	$ 540,226	$56,135	$ 484,092
18 (Hybrid)	3	2	2	7	$ 480,862	$35,436	$ 445,426
19 (Hybrid)	3	3	3	9	$ 521,139	$35,436	$ 485,703
20 (Hybrid)	3	2	3	8	$ 549,467	$35,436	$ 514,031

APPENDIX E

FACTSHEET FOR THE SOUTHEAST CLIMATE SCIENCE CENTER: TURNING UNCERTAINTY INTO ACTIONABLE INFORMATION FOR CONSERVATION DECISIONS

This factsheet is a draft that has not been submitted to or approved by the Southeast Climate Science Center or US Geological Survey.

TURNING UNCERTAINTY INTO USEFUL INFORMATION FOR CONSERVATION DECISIONS

DEPARTMENT *of the* INTERIOR
SE CLIMATE SCIENCE CENTER
SE CSC Fact Sheet 2015-02
http://globalchange.ncsu.edu/secsc

In a nutshell...

Uncertainty is ***Pervasive***.

Perceptions of uncertainty vary.

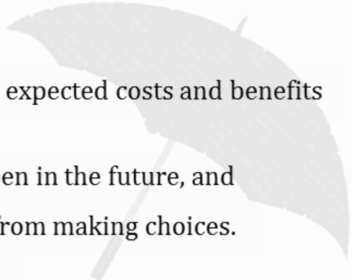
Progress can be made despite uncertainty.

Uncertainty is always present in conservation and other socio-ecological decisions, which can make choices uncomfortable and challenging. All choices have consequences – including the choice to do nothing. This fact sheet discusses the *pervasiveness* of uncertainty, the importance of understanding varying *perceptions* of uncertainty, and avenues for *progress* in the presence of uncertainty and differing risk tolerances.

> *Our goal is to support deliberate conservation decisions that are robust to unavoidable societal and ecological uncertainties.*

The Umbrella Example

It's 6:00am. You look, bleary-eyed, out the window to see a calm sky with some clouds in the distance. You get up, get dressed, and prepare for the day. You pack your bag, grab your coffee, and head towards the door. You pause at the sight of your umbrella. Should you bring it? You quickly weigh the options of unnecessarily carrying the umbrella all day versus potentially getting rained on...

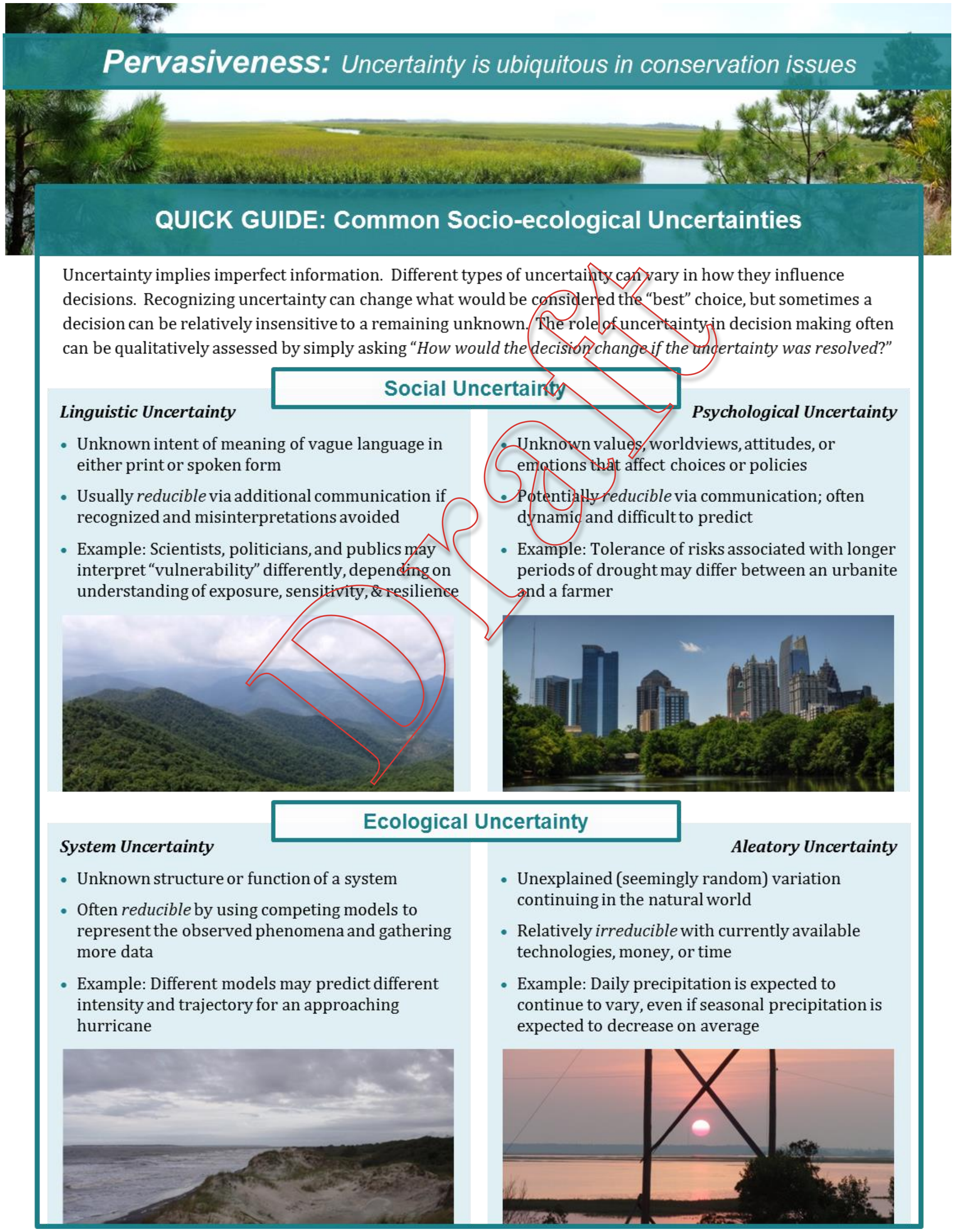


Three things should be clear from this familiar scenario:

1) we regularly face choices that require us to weigh the expected costs and benefits associated with different options,
2) rarely are we absolutely certain about what will happen in the future, and
3) imperfect information, by itself, does not prevent us from making choices.

We move forward and take on the day. Sometimes it storms.

Factsheet page 1



Pervasiveness: *Uncertainty is ubiquitous in conservation issues*

QUICK GUIDE: Common Socio-ecological Uncertainties

Uncertainty implies imperfect information. Different types of uncertainty can vary in how they influence decisions. Recognizing uncertainty can change what would be considered the "best" choice, but sometimes a decision can be relatively insensitive to a remaining unknown. The role of uncertainty in decision making often can be qualitatively assessed by simply asking "*How would the decision change if the uncertainty was resolved*?"

Social Uncertainty

Linguistic Uncertainty

- Unknown intent of meaning of vague language in either print or spoken form
- Usually *reducible* via additional communication if recognized and misinterpretations avoided
- Example: Scientists, politicians, and publics may interpret "vulnerability" differently, depending on understanding of exposure, sensitivity, & resilience

Psychological Uncertainty

- Unknown values, worldviews, attitudes, or emotions that affect choices or policies
- Potentially *reducible* via communication; often dynamic and difficult to predict
- Example: Tolerance of risks associated with longer periods of drought may differ between an urbanite and a farmer

Ecological Uncertainty

System Uncertainty

- Unknown structure or function of a system
- Often *reducible* by using competing models to represent the observed phenomena and gathering more data
- Example: Different models may predict different intensity and trajectory for an approaching hurricane

Aleatory Uncertainty

- Unexplained (seemingly random) variation continuing in the natural world
- Relatively *irreducible* with currently available technologies, money, or time
- Example: Daily precipitation is expected to continue to vary, even if seasonal precipitation is expected to decrease on average

Factsheet page 2

Perceptions: *Uncertainty varies in how it is interpreted and used*

Treatment of Uncertainty

Why do people who share a common goal make different choices, even when presented with identical information?

- Our individual and collective responses to uncertainty are likely shaped by our tolerance to risk; attitudes can range from risk-avoiding to risk-seeking.
- Individual worldviews or values, which may be shaped by one's psychology, experience, education, culture, politics, or religion, may further influence how different types of uncertainty bear upon a decision and can even be dynamic for an individual decision maker.

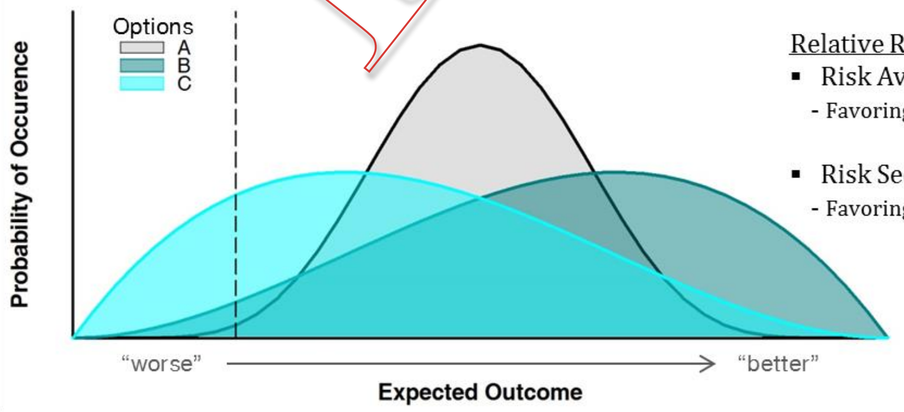
Uncertainty is ever present, and its existence may be used as the reason for 1) doing less, 2) maintaining the status quo, or 3) doing more.

When we view uncertainty from a decision-analysis perspective, the important questions become: *How was the decision justified?, What risks were treated as acceptable?, & Whose points of view about risk were considered?*

Applying a Decision-Making Perspective to Uncertainty & Risk

For decision makers, the perceived "*risk*" associated with making a choice can frequently be considered as a combination of the probability of experiencing an outcome and the severity of the outcome itself.

Consequence Distributions for a Decision Based upon Imperfect Information



Relative Risk Tolerances

- Risk Avoiding
  - Favoring option A over B
- Risk Seeking
  - Favoring option C over A

Example output from a decision analysis – Uncertainty leads to distributions of potential outcomes resulting from implementation of three alternative options (A, B, and C). Curve height indicates the probability of observing an outcome if the decision alternative is implemented, and a dashed vertical line represents a minimum threshold, below which we would be considerably unhappy with the outcome of the decision. Option A has a moderate expected value, relatively low uncertainty, and the lowest probability of experiencing conditions below the threshold. Option B has the highest expected value, but a higher probability of an very undesirable outcome than A. Option C has the lowest expected value, but a higher probability of a very desirable outcome than A.

Progress: *Uncertainty is not necessarily a roadblock for science or policy*

Key Steps for Producing Actionable Science & Important Questions to Confront

1) **Specify objectives –** What do we hope to achieve?

2) **Identify options –** What can we do about it?

3) **Make predictions –** What are the likely consequences?

4) **Evaluate uncertainties and tradeoffs –** What are the risks?

Actionable Science: Uncertainty → Useful Information

4 Propositions for the Co-Production of Actionable Science

I. **Structure decisions –** A decision-making perspective can help stakeholders, scientists, and decision makers navigate the cooperative pursuit of actionable science. Purposefully putting knowledge into practice often requires: 1) specification of objectives that are driving the decision, 2) identification of policy options that may be able to achieve these objectives, 3) formal assessment of the anticipated consequences of alternative options, and 4) evaluation of the risks of different courses of actions given unavoidable uncertainty. Formal evaluation of what is known and unknown leads to better informed and more justifiable decisions.

II. **Establish Roles –** Explicitly recognizing how both scientific information and societal values affect conservation decisions will allow accepted risks to be more transparent and paths towards progressively well-informed future choices to be more evident. Effective science-policy partnerships at the knowledge-action interface should allow for efficient production of useful information. In order for scientists and policymakers to cooperatively pursue actionable science, they will need to share an understanding of the best available information, how to acquire new decision-relevant information, and the feasibility of alternatives.

III. **Recognize how values motivate –** The scientific process is designed to confront uncertainty, but uncertainty about the future will always remain. Even if perfect knowledge was obtainable, decisions would remain defined by value-based objectives.

IV. **Heed evidence –** Actionable science focuses on the reduction of decision-relevant uncertainty, but it does not tell decision makers what choice to make. A transparent, actionable science process can characterize the impacts of uncertain information on a decision and allow ethical discussions about decision objectives and alternative to occur apart from the scientific enterprise. The quality of a decision will often be judged based upon how available information was used & what efforts were made to increase knowledge for the future. Learning can have costs, but so can doing nothing.