A REGIONAL SYNTHESIS OF CLIMATE DATA TO INFORM THE 2025 STATE WILDLIFE ACTION PLANS IN THE NORTHEAST U.S.

2024

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EXECUTIVE SUMMARY

Authors: Michelle Staudinger, Karen Terwilliger

The State Wildlife Action Plans (SWAPs) are proactive planning documents, known as “comprehensive wildlife conservation strategies,” that assess the health of each state’s wildlife and habitats, identify current management and conservation challenges, and outline needed actions to conserve natural resources over the long term. SWAPs are revised every 10 years, with the last revision in 2015 and the next revision anticipated in 2025. State managers have a long history of managing for threats such as land-use change, pollution, and harvest. However, they have expressed a lack of expertise and capacity to keep pace with the rapid advances in climate science and noted that much of the information available is not scaled to meet their needs; thus making the prospect of integrating climate information into SWAPs a daunting task. (Johnson, 2018; Yocum et al., 2021; Blandford, 2022).

This report, led by the Northeast Climate Adaptation Science Center (NE CASC), directly addresses SWAP climate science data needs through a Northeast regional synthesis across four key areas of climate science: 1) observed and projected climate changes, 2) species responses to climate change, 3) climate vulnerabilities and risks, and 4) scale-specific adaptation strategies and actions. In addition, case studies of climate adaptation efforts and climate threat-to-action narratives provide illustrative examples of how climate change frameworks and tools are being operationalized in decision-making processes related to Regional Species of Greatest Conservation Need (RSGCN) and their habitats across the region. Lists of recent climate resources were also synthesized into extensive data tables to provide SWAP writing teams with a comprehensive platform of information to support content development (AFWA 2022).

The current synthesis builds on a similar effort that was delivered in 2015 (Staudinger et al., 2015) and nearly a decade of work since, as developed by numerous partners and collaborative networks across the region. Starting in 2020, the NE CASC funded three projects to support state engagement and information exchange (Assessing climate change threats and actions in NE SWAPs, led by Terwilliger Consulting Inc.), the development of reliable and consistent regional and state-specific climate projections (Enhancing the reliability and usability of climate change information for WAPs in the NE U.S., led by Ambarish Karmalkar), and provide research capacity by appointing three ORISE fellows (A regional synthesis of climate...
data to inform the 2025 SWAPs in the NE U.S., led by NE CASC staff). Once established, these three teams began formal planning for this report in close coordination with the Northeast Association of Fish and Wildlife Agencies (NEAFWA) SWAP coordinators, and the Northeast Fish and Wildlife Diversity Technical Committee (NEFWDTC). The synthesis scope was focused through regular meetings with state personnel and monthly discussions of the Northeast Climate Change Working Group, a forum established to solicit and provide specific input and updates on regional adaptation activities in the Northeast. The final product is the result of a four-year iterative process that provided states with multiple platforms and opportunities to receive progress updates and provide feedback to NE CASC project teams as synthesis content was developed.

The current report is distinct compared to other climate assessments in that it provides information that has been scaled specifically to the 13 state footprint of the NEAFWA region, and focused on addressing the unique threats to 418 Northeast RSGCN. The most recent RSGCN list, finalized in 2023 (TCI & NEFWDTC, 2023), was used to search for studies of species responses to climate change (Chapter 2) and climate vulnerability rankings (Chapter 3). While the majority of information used in the report was derived from recent resources published since 2015, in some cases, cumulative records were summarized to facilitate the use of the total breadth of information available; for example, the appendices associated with Chapter 3 on Climate Change Vulnerability Assessments results contain all rankings assembled from Staudinger et al., (2015) through those released in 2023.

In reflecting on what has changed since the last regional synthesis report and what is new in this version; first, the spatial range of the information has narrowed from the Midwest and Northeast regions to just the Northeast region to align with the shifting footprint of the NE CASC. In 2015, climate data were only provided at the regional level, whereas in the current report climate data and projections are summarized at the state, ecoregion, and NEAFWA scales. Model selection for the climate projections were selected based on agreement between observed historic trends over the Northeast region and provide robust information for states to use. Because the same models, scenarios, and downscaling methods were used to project trends for each state and across the region, the states can use the information in this report to make local and coordinated landscape-scale decisions based on a common foundation of climate data. Second, the timing of the delivery of the current report is much earlier in the SWAP
development process this time, being delivered at the start of report writing rather than at the end as was true in 2015. This earlier timeline was the direct result of established relationships having been developed during the production of the 2015 report (Staudinger et al., 2015), as well as the maturity of the science agenda, portfolio, and staff of the NE CASC over the last 10 years. Finally, in terms of content, the amount, depth, and breadth of climate science available is substantially greater than a decade ago. For example, the contents of the current report are enhanced by spatially explicit downscaled climate projections, thousands of new studies of species’ biological responses to climate variables, and hundreds of new climate vulnerability rankings. Perhaps most notable is the shift from theory to implementation of climate adaptation strategies and actions that have come to fruition over the last decade. In the 2015 report, the content was limited to conceptual descriptions of emerging adaptation frameworks and philosophies; in contrast, there are now numerous case studies demonstrating how climate adaptation actions have been operationalized across multiple scales with first-hand narratives from natural resource managers of their learned experiences applying novel adaptation strategies in the field.

The content contained in this decadal update is substantial. To make the content approachable and accessible, information is organized in sections that align with the eight required SWAP elements and the voluntary guidance on climate adaptation from the Association of Fish and Wildlife Agencies (AFWA, 2022). In Chapter 1, we provide a suite of climate projections scaled to the Northeast region to align with SWAP Element 4. In Chapter 2, we present recent research on how RSGCN and their associated habitats are responding to climate impacts (e.g., through shifts in distribution, abundance, location, and condition) and identify knowledge gaps to align with SWAP Elements 1, 2, and 3. In Chapters 3 and 4, we synthesize new climate vulnerability assessments, adaptation frameworks, strategies, and actions to inform conservation objectives and monitoring targets to align with SWAP Elements 3, 4, and 5. Finally, the case studies and examples of implemented adaptation initiatives in Chapters 4 and 5 provide illustrations of how State agencies can work with partners and involve the public to align with SWAP Elements 7 and 8. We envision that depending on their area of expertise and focus, SWAP authors can review the main text of each chapter to get a broad understanding of the state-of-the-science and the options available to them and then use the data tables and appendices to find context-specific information and tools from reference lists and links contained within.
Chapters 3 and 4 provide step-by-step guidance on how to interact with the comprehensive lists of climate vulnerability rankings and adaptation action menus to find information to prioritize conservation targets and develop specific strategies to reduce climate and multi-stressor risks.

Overall, the findings detailed in the report show that the Northeast is warming and sea levels are rising at rates that exceed the global average, making the region a climate change hotspot (Chapter 1). Climate impacts are not spatially or seasonally uniform across the region, making downscaled projections important to understanding how conditions will manifest differently across the region for climate vulnerable RSGCN and associated habitats. Extreme events, which were not long ago considered something to plan for in the future, are now impacting the Northeast on a regular basis. Extreme events take many forms in the Northeast, including heat waves, coastal flooding – even on sunny days, storms, drought, and pest outbreaks. Storm events in particular, whether they take the form of rain, ice, or wind from larger episodic systems such as Nor’easters or localized seasonal precipitation events, are news-worthy for their impacts on human infrastructure and communities, but these events are also causing significant impacts on the habitats and underlying ecological processes that support RSGCN. Therefore, the continued trajectory of climate change in all forms is critical to consider as managers engage in resilience and adaptation planning. The content in this report is intended to assist states to explicitly and deliberately speak to the climate risks and adaptation options when developing SWAP actions. SWAP authors will find in the chapters of this report:

- State-specific climate summaries for all 13 states in the NEAFWA footprint (Chapter 1)
- A synthesis of over 450 new articles on current and proposed RSGCN (Chapter 2)
- 24 new Climate Change Vulnerability Assessments with 392 new records for NE RSGCN and their associated habitats, including 92 species not previously captured in CCVAs (Chapter 3)
- 9 "big picture" climate adaptation decision support framework, 16 new Climate Adaptation Action Menus, seven summaries of emerging climate adaptation topics, and 81 additional climate adaptation resources (Chapter 4)
- 4 new illustrative climate adaptation case studies (Chapter 5)

Climate models identify ~2035-2040 as the critical point in time when projections begin to diverge towards very different future scenarios of more moderate (e.g., Representative Concentration Pathway (RCP) 4.5) or aggressive changes (e.g., RCP 8.5) (IPCC 2023).
Consequently, the actions taken over the next 10-15 years are critical for management to put adaptation programs in place to monitor and prepare for the greatest climate changes experienced to date and yet to come. Prioritizing research and monitoring to reduce uncertainty and directly address knowledge gaps for species with limited or no climate response information and those that have not yet been evaluated through Climate Change Vulnerability Assessments are key climate adaptation actions for SWAPs to implement in the near-term. In addition, more nuanced actions may be considered for species that have already been identified as highly climate vulnerable before greater climate impacts are realized. The good news is that the number of climate adaptation resources has grown substantially in recent years and managers have many new options to create innovative climate-informed conservation responses (Stein et al., 2024). Many of the conservation strategies that state managers are already familiar with such as reducing the influence of non-climate stressors by protecting populations and habitats from overharvest, pollution, fragmentation and other anthropogenic activities have been shown to be effective in increasing resilience to climate change. Incorporating forward-looking climate projections and data into management is facilitated by the breadth of research and knowledge that is now readily available. The primary aim of this report is to synthesize the best available climate science and provide an accessible foundation of information to empower SWAP authors to use in meeting the challenge that climate change brings to their decadal planning process.

LITERATURE CITED


CHAPTER 1: CLIMATE CHANGE IN THE NORTHEAST UNITED STATES

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Key findings

● The Northeast U.S. (NEUS) has warmed by 1.4°C (2.5°F) and has experienced a steady increase in total annual precipitation since 1895. Climate projections presented in this report indicate a continuation of these observed trends in the future.

● Coastal regions of the NEUS show pronounced increases in temperature and extreme precipitation over the historical period. On the other hand, historical temperature change in the interior NEUS has been weaker in comparison, creating a spatially heterogeneous pattern of warming.

● Average and extreme temperatures are projected to increase but the exact magnitude of the increases is dependent on the future trajectory of greenhouse gas emissions.
• Average and extreme precipitation are projected to increase this century but high variations in year-to-year projections remain. Due to high variability, precipitation projections for different emissions scenarios overlap completely throughout this century, indicating more attention should be paid to precipitation variability than any particular emissions scenario.

• Climate projections reported here are consistent with those reported previously, providing reliable trajectories for risk assessment and management decision-making.

I. INTRODUCTION

The Northeast United States (NEUS) is vulnerable to a range of climate impacts including heatwaves, heavy precipitation, and sea level rise with serious consequences for natural and human systems and resources, Indigenous Peoples within the region, infrastructure, and tourism (Whitehead et al., 2023). Global average temperature has increased by about 1.2°C (2.1°F) since the industrial revolution (IPCC, 2023) in response to increasing greenhouse gas (GHG) concentrations. Global warming will continue to increase in the near term and will likely reach 1.5°C (2.7°F) irrespective of the emissions scenario (IPCC, 2023). A best estimate for global warming by the end of this century is 2.7°C (4.9°F), ranging from 1.4°C (2.5°F) for a very low GHG emissions scenario to 4.4°C (7.9°F) for a very high emissions scenario.

The NEUS is situated in the mid-latitude westerlies zone. This means that the region, despite its proximity to the Atlantic Ocean, experiences a continental climate characterized by warm summers and cold winters, with most weather systems moving in from the west (Zielinski and Keim, 2005). In coastal regions of the NEUS, however, temperature and precipitation extremes are also impacted by conditions over the Atlantic Ocean (e.g., coastal storms). The NEUS climate, in general, exhibits high seasonal and year-to-year (interannual) variations due to complex interactions between regional characteristics (e.g., topography, coastal geography) and large-scale interactions between local and hemispheric-scale atmospheric circulation. This has resulted in several bioclimatic or ecological zones across the Northeast.
Figure 1.1: (a) The Northeast Association of the Fish and Wildlife Agencies (NEAFWA) states in the Northeast U.S. covered in this chapter (Figure source: Northeast Regional Conservation Needs) and (b) EPA Level III ecoregions. Different colors indicate distinct ecoregions across the NEUS. Further information about these ecoregions can be found on the EPA website.

The synthesis and analysis described in this chapter provide climate change information for the northeastern region encompassing the thirteen states that are part of the Northeast Association of Fish and Wildlife Agencies (NEAFWA; Fig. 1.1a). The information is created to inform the State Wildlife Action Plans (SWAP) of the NEAFWA states. Quantitative climate projections are also delineated using the EPA Level III ecoregion definitions for the Northeast as shown in Figure 1.1b. The EPA ecoregions denote similarities in their biotic, abiotic, terrestrial, and aquatic ecosystem components (Omernik and Griffith, 2014) and can be considered appropriate spatial scales for aggregating climate data. While management decisions typically consider the next 5- to 20-year planning horizons, we provide projections for both the near-term as well as end-of-century to offer a long-term perspective on anticipated changes.

II. CLIMATE DATA

Climate observations over the historical period described in this report are obtained from the National Atmospheric and Oceanic Administration (NOAA) website. The observed daily temperature and precipitation data are part of the nClimDiv dataset (Karl and Koss, 1984; Voss et
which is based on the Global Historical Climatology Network Daily (GHCND) database (Menne et al., 2012; Durre et al., 2010).

The climate change projections included here are based on climate models from a large ensemble of standardized model simulations (called the Coupled Model Intercomparison Project Phase 5; CMIP5) and two pathways of future greenhouse gas concentrations (called Representative Concentration Pathways; RCP; Meinshausen et al. 2011): RCP4.5 and RCP8.5, the medium and high emissions scenarios, respectively. Climate models were carefully selected from the CMIP5 archive based on their ability to simulate the historical climate of the NEUS while maintaining diversity in future projections consistent with known uncertainties (Karmalkar et al. 2019). A better agreement between climate model simulations and observations over the historical period increases our confidence in the models’ ability to provide reliable projections and a diverse set of projections provides a range of plausible outcomes for risk assessment. We used the statistically downscaled counterparts (called MACAv2-METDATA downscaling; Abatzoglou and Brown, 2012) of the selected climate models to generate data for variables of interest at state, ecoregion, and watershed levels to aid risk assessments and management decisions. The MACAv2 dataset provides daily data for temperature and precipitation and is gridded at the 4km (1/24°) resolution.

What we describe in this report are climate projections and not predictions. It is important to note the key difference between these terms and how they get used to describe results obtained from climate models. Projections show a range of what could happen based on a number of future scenarios. In contrast, predictions describe what is anticipated when one particular scenario plays out. Model projections (i.e., what could happen) are not predictions (i.e., what will happen) because the outcome depends on how greenhouse gas emissions change over time as policies and human activities shift, aspects that are represented by different emissions scenarios. The combination of multiple models and two emissions scenarios maximizes the likelihood that the projections for all variables bracket the range of plausible future outcomes. Carrying out risk assessments for all presented climate futures and presenting a range of vulnerability scores can for example help triage species that may be sensitive to any amount of climate change, versus those that may only be sensitive to a worst-case scenario.

Climate change over the historical and future periods are presented in this report primarily in two graphical formats: The plume plots show the time evolution in the absolute
values for a given variable, and the maps depict the spatial pattern of projections for a selected future period. To create maps and to provide quantitative projections, we select two 30-year periods in the future: 2030-2059 and 2070-2099. The projections for these periods are calculated relative to 1990-2019, which we consider as the ‘baseline’ or the ‘present-day’ climate.

III. OBSERVED AND PROJECTED CLIMATIC CHANGES

A) TEMPERATURE

The observations show a clear warming trend between 1895 and 2022 (Fig. 1.2a) in the NEUS, reflecting an overall warming of 1.4°C (2.5°F) since 1895. Over the same period, the global average temperature has increased by about 1.1°C (2°F). Higher warming in the NEUS compared to the global average is expected since land warms faster than the oceans. This warming trend is projected to continue in the future, as elaborated later. Note that the observed warming in the NEUS is not spatially uniform across the region. Specifically, the coastal regions of the NEUS, from Maine to New Jersey/Delaware, have experienced enhanced warming compared to the interior parts west of the Appalachians. This pattern of warming is present in all seasons but is more pronounced in summer months. In fact, the coastal NEUS and the adjacent Northwest Atlantic continental shelf (NWAS) regions have concurrently emerged as climate change hotspots as highlighted in studies by Pershing et al. (2015, 2021) and Karmalkar and Horton (2021).

The spatial pattern of observed warming depicted in Fig. 1.2b underscores the exceptional nature of coastal warming in the NEUS, especially when compared to the other parts of the eastern U.S. Investigation into mechanisms responsible for the observed exceptional long-term coastal warming in the NEUS hints at the interplay between changes in ocean and atmospheric circulation in the North Atlantic region (Karmalkar and Horton, 2021).
Figure 1.2: (a) The annual mean surface air temperature in the Northeast U.S. between 1895 and 2022. The black line shows the twentieth-century average, and the blue line shows the trend over the entire period. (Source: NOAA Climate at a Glance) (b) Temperature change over the historical period across the contiguous US. (Source: NCA5; Marvel et al., 2023)

Consistent with the observed trend in mean temperature, both minimum and maximum temperatures in the region have risen over the historical period. Figures 1.3a and 1.3b illustrate the yearly variations and trends in annual (365-day averaged) maximum and minimum temperatures in the NEUS. Over the period from 1895 to 2022, both maximum and minimum temperatures exhibit a steady upward trend. These changes reflect region-wide increases in both daytime high temperatures and night-time low temperatures.

Figure 1.3: The annual mean (a) maximum and (b) minimum temperature in the Northeast U.S. between 1895 and 2022. The black line shows the twentieth-century average, and the blue and red lines in a and b, respectively, show trend lines over the entire period. (Source: NOAA Climate at a Glance)
The future increases in regional temperatures will depend on the future emissions trajectories (Fig. 1.4). The annual mean temperature by the end of this century is projected to reach about 55°F in a medium emissions scenario, RCP4.5, and about 60°F in a high emissions scenario, RCP8.5, increasing by about 5-10°F above the average over the recent three decades (1990-2019). The future warming in the NEUS is not sensitive to the emissions scenarios until the middle of this century, with all scenarios following the same general trajectories until the 2040s and diverging thereafter.

**Figure 1.4:** Projected changes in the annual (ANN) average temperature. (a) the end-of-century (2071-2100 mean) projections relative to the present day (1991-2020) mean. The projections show mean values across the nine climate models for the medium emissions scenario, RCP4.5. (b) The historical and projected temperature in °F as simulated by climate models. Blue and red lines show median projections under medium and high scenarios, RCP4.5 and RCP8.5, respectively. The shading indicates the spread in projections across nine climate models and all gridboxes within the region. The time series projections shown in b were calculated across all gridboxes (and not for spatial averages) to capture the spatial variability and indicate the minimum and maximum projections possible within the region.
Table 1.1: Northeast and state-level projections (ordered north to south) for annual mean surface air temperature for two periods in the future under the medium emissions scenario, RCP4.5. The 5th to 95th percentile spread in projections is based on nine climate models and spatial variability within each region.

<table>
<thead>
<tr>
<th>Region</th>
<th>Average Temperature [ANN]</th>
<th>Baseline (°F)</th>
<th>Projections (RCP45, change in °F)</th>
<th>Late-century (2070-2099)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>(1990-2019)</td>
<td>5th</td>
<td>50th</td>
</tr>
<tr>
<td>Northeast</td>
<td></td>
<td></td>
<td>50.43</td>
<td>+1.2</td>
</tr>
<tr>
<td>Maine</td>
<td></td>
<td></td>
<td>42.55</td>
<td>+1.4</td>
</tr>
<tr>
<td>New Hampshire</td>
<td></td>
<td></td>
<td>45.02</td>
<td>+1.2</td>
</tr>
<tr>
<td>New York</td>
<td></td>
<td></td>
<td>47.04</td>
<td>+1.3</td>
</tr>
<tr>
<td>Vermont</td>
<td></td>
<td></td>
<td>43.90</td>
<td>+1.3</td>
</tr>
<tr>
<td>Massachusetts</td>
<td></td>
<td></td>
<td>49.24</td>
<td>+1.2</td>
</tr>
<tr>
<td>Connecticut</td>
<td></td>
<td></td>
<td>50.52</td>
<td>+1.2</td>
</tr>
<tr>
<td>Rhode Island</td>
<td></td>
<td></td>
<td>51.32</td>
<td>+1.2</td>
</tr>
<tr>
<td>Pennsylvania</td>
<td></td>
<td></td>
<td>50.05</td>
<td>+1.2</td>
</tr>
<tr>
<td>New Jersey</td>
<td></td>
<td></td>
<td>54.17</td>
<td>+1.1</td>
</tr>
<tr>
<td>Maryland</td>
<td></td>
<td></td>
<td>56.43</td>
<td>+1.1</td>
</tr>
<tr>
<td>Delaware</td>
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<td></td>
<td>57.23</td>
<td>+1.0</td>
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<tr>
<td>Virginia</td>
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<td>56.73</td>
<td>+1.1</td>
</tr>
<tr>
<td>West Virginia</td>
<td></td>
<td></td>
<td>52.95</td>
<td>+1.2</td>
</tr>
</tbody>
</table>

An increase in heat extremes is one of the obvious consequences of rising temperatures. One indication of the expected increase in heatwaves is the steady rise in annual mean maximum temperature (daytime highs) in the NEUS over the last century. We calculate the change in this metric by examining the number of days per season or per year that cross the maximum daily temperature thresholds of 90°F, 95°F, and 100°F. For example, Figure 1.5 shows the spatial pattern of change in days > 95°F by the end of this century (2071-2100 mean) relative to the present day (1991-2020 mean). In the medium emissions scenarios, the entire region will experience an increase in the number of hot days.
Figure 1.5: Projected changes in the number of days with maximum temperature, $T_{\text{max}}$, above 95°F. (a) The end-of-century (2071-2100 mean) projections relative to the present day (1991-2020) mean. To indicate the highest values plausible for this variable, the projections show maximum values at every gridbox across the nine climate models for the medium emissions scenario, RCP4.5. (b) The historical and spatially projected values for the number of days with $T_{\text{max}}$ above 95°F as simulated by climate models. Blue and red lines show median projections under medium and high scenarios, RCP4.5 and RCP8.5, respectively. The shading indicates the spread in projections across nine climate models and all gridboxes within the region. The time series projections shown in b were calculated across all gridboxes (and not for spatial averages) to capture the spatial variability and indicate the minimum and maximum projections possible within the region.

Figure 1.5 also highlights that the projections for hot extremes show regional variations across different ecoregions in the southern part of the study domain. In Table 1.2, we summarize the mean increase in the number of days above 95°F for seven different ecoregions ranging from coastal plains to the interior plateau. Compared to the high-elevation Central Appalachians ecoregion, the coastal ecoregion of Piedmont experiences significantly more hot days per year. More generally, the projections also indicate higher increase in the number of hot days in coastal (e.g., Piedmont, Middle Atlantic Coastal Plain) and interior (e.g., Western Allegheny Plateau) ecoregions compared to the mountainous middle ecoregions (e.g., Central Appalachians, Ridge and Valley).
Table 1.2: Ecoregion-level projections for the number of days per year above 95°F for two periods in the future under the high emissions scenario, RCP8.5. The 5th to 95th percentile spread in projections is based on 9 climate models for area averaged values for each region.

B) PRECIPITATION

The NEUS receives abundant and relatively uniform precipitation throughout the year, but there can be large variations from one year to the next. The region has experienced a modest increase in total annual precipitation (Marvel et al., 2023; Easterling et al., 2017) with a relatively strong increasing trend in the warm season, and with a dramatic increase in very heavy rainfall (top 1% of events; Whitehead et al., 2023; Hoerling et al., 2016; Wuebbles et al., 2017) over the last 60 years. Figure 1.6a shows a steady increasing trend in annual mean precipitation since 1895 in the NEUS. The 1970s and 2000s also feature multi-year periods during which the region experienced significantly wetter conditions than the 20th century average. The spatial pattern of observed precipitation change (Fig. 1.6b) highlights the increase in precipitation in every part of the NEUS, in line with precipitation increases experienced in the eastern half of the United States. There are clear indications that its distribution throughout the year has also changed. For one, seasonal changes in precipitation in the last three decades relative to the first half of the twentieth century indicate that the wetting trend in the NEUS is pronounced in summer (June, July, August; JJA) and fall (September, October, November; SON), and is weakest in winter.
A significant portion of the wetting trend in summer and fall is related to an increase in the intensity of heavy precipitation events related to tropical (Barlow, 2011) and extratropical storms (e.g., Nor’easters, Kunkel et al., 2012). Many extreme precipitation indices—including the 5-year maximum precipitation, and the total precipitation falling in the top 1% of all days with precipitation—indicate increases since 1901 with substantial increases in the last five decades (Easterling et al., 2017). Other indices such as the number of 2-day extreme precipitation events also show large increases over the eastern half of the U.S. (Easterling et al., 2017). This trend of increasing mean and extreme precipitation is projected to continue with warming in the future (Maloney et al., 2014; Rawlins et al., 2012; Fan et al., 2015; Ning et al., 2015).
Figure 1.7: Projected changes in the total annual precipitation. (a) The end-of-century (2071-2100 mean) projections, in inches, relative to the present day (1991-2020) mean. The projections show mean values across the nine climate models for the medium emissions scenario, RCP4.5. (b) The historical and projected precipitation in inches as simulated by climate models. Blue and red lines show median projections under medium and high scenarios, RCP4.5 and RCP8.5, respectively. The shading indicates the spread in projections across nine climate models and all gridboxes within the region. The time series projections shown in b were calculated across all gridboxes (and not for spatial averages) to capture the spatial variability and indicate the minimum and maximum projections possible within the region.

Consistent with the observed wetting trend, most climate models suggest a steady increase in future precipitation in the region (Fig. 1.7). The trend, however, is small compared to high seasonal and interannual variability in precipitation. High variability means that despite an overall wetting in the future, the region will remain susceptible to short-term drought conditions. Indeed, despite an overall increase in precipitation in recent decades, the region has experienced both short-term (seasonal, e.g., 2016) as well as long-term (multi-year, e.g., 1960s, early 2000s) droughts with significant impacts on human and natural systems (Peterson et al., 2013). A case study of how drought has led to cascading effects in forest ecosystems in the NEUS is discussed in Chapter 5.
Table 1.3: Northeast and state-level projections of annual total precipitation for two periods in the future under the medium emissions scenario, RCP4.5. The 5th to 95th percentile spread in projections is based on 9 climate models and spatial variability within the region under consideration.

<table>
<thead>
<tr>
<th>Region \ Percentiles</th>
<th>Baseline (1990-2019)</th>
<th>Projections (RCP4.5, change in inches)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>5th</td>
<td>50th</td>
</tr>
<tr>
<td>Northeast</td>
<td>43.76</td>
<td>-1.7</td>
</tr>
<tr>
<td>Maine</td>
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<td>+1.4</td>
</tr>
<tr>
<td>New Hampshire</td>
<td>47.23</td>
<td>+0.5</td>
</tr>
<tr>
<td>New York</td>
<td>42.72</td>
<td>-0.5</td>
</tr>
<tr>
<td>Vermont</td>
<td>45.88</td>
<td>-0.0</td>
</tr>
<tr>
<td>Massachusetts</td>
<td>48.91</td>
<td>-0.3</td>
</tr>
<tr>
<td>Connecticut</td>
<td>50.86</td>
<td>+0.6</td>
</tr>
<tr>
<td>Rhode Island</td>
<td>50.04</td>
<td>-0.0</td>
</tr>
<tr>
<td>Pennsylvania</td>
<td>43.66</td>
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</tr>
<tr>
<td>New Jersey</td>
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<td>-0.7</td>
</tr>
<tr>
<td>Maryland</td>
<td>44.59</td>
<td>-2.1</td>
</tr>
<tr>
<td>Delaware</td>
<td>45.03</td>
<td>-1.3</td>
</tr>
<tr>
<td>Virginia</td>
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<td>-2.4</td>
</tr>
<tr>
<td>West Virginia</td>
<td>45.81</td>
<td>-4.2</td>
</tr>
</tbody>
</table>

Another important aspect of local-scale precipitation projections is that the projections are not sensitive to future emissions pathways. Projections for both medium and high emissions pathways (red and blue plumes in Fig. 1.7) overlap completely throughout this century, which indicates that more attention should be paid to the full range of precipitation variability than any particular emissions scenario. Additionally, given the large variability in precipitation, it is important to understand how extreme precipitation including both heavy precipitation events and dry conditions are projected to change.
Figure 1.8: Projected changes in the number of days with daily precipitation above 1 inch. (a) The end-of-century (2071-2100 mean) projections relative to the present day (1991-2020) mean. To indicate the highest values plausible for this variable, the projections show maximum values at every gridbox across the nine climate models for the medium emissions scenario, RCP4.5. (b) The historical and projected values for the number of days with daily precipitation above 1 inch as simulated by climate models. Blue and red lines show median projections under medium and high scenarios, RCP4.5 and RCP8.5, respectively. The shading indicates the spread in projections across nine climate models and all gridboxes within the region. The time series projections shown in b were calculated across all gridboxes (and not for spatial averages) to capture the spatial variability and indicate the minimum and maximum projections possible within the region.

C) ATMOSPHERIC MOISTURE

As temperatures rise, the capacity of the atmosphere to hold water vapor increases. Additionally, increases in sea surface temperatures can also lead to more evaporation, which puts more water vapor in the atmosphere. Consequently, in general, there is a tendency for higher atmospheric water vapor content under warmer conditions (Held and Soden, 2006; Boer 1993). With increased atmospheric moisture, there is an elevated risk of extreme precipitation events (Trenberth, 1999; Pall et al., 2007; Pendergrass, 2018), including severe storms and heavy rainfall. The projected increase in the number of days with heavy precipitation described above is a consequence of the increased moisture content in the future. A combination of warming and increased humidity also has serious consequences for human health. The presence of high humidity during future heat waves will reduce the ability of the human body to cool via evaporation and can cause severe heat stress and mortality (Rastogi et al., 2020).
Evapotranspiration is a combination of two processes: first, water evaporation from the oceans, inland water bodies, and land surface, and second, transpiration from vegetation. While evapotranspiration is expected to increase with warming in the global sense, regionally it is influenced by local climate conditions and vegetation and land cover type. For instance, transpiration is dominant in forested landscapes such as the NEUS (e.g., Zhang et al., 2016) and is controlled by moisture content in the local atmosphere, vegetation, and soil. Recent research shows regional variations in evapotranspiration across the NEUS. Since the 1940s, evapotranspiration has increased in the cooler northern parts of NEUS while it has decreased in some of the relatively warmer southwestern parts of the NEUS (Vadeboncoeur et al., 2018).

D) STORMS

Storms in summer and fall: Large storms originating in the tropics, referred to as hurricanes in the Atlantic Ocean, bring abundant moisture and heavy rainfall to the NEUS. The impact of ocean and atmosphere warming on the frequency and intensity of hurricanes is an active area of research. While there is no long-term trend in the frequency of landfalling hurricanes in the smaller domain of the U.S. (Marvel et al., 2023), recent research does suggest a significant increase in the rapid intensification of hurricanes due to anthropogenic warming (Bhatia et al., 2022).

Tropical storms can cause significant wind damage along the coast as well as flooding across the NEUS, which is compounded by rising sea levels (see Douglas and Kirshen, 2022 and references therein). The NEUS is especially vulnerable to tropical storms moving northward as they increase in size considerably while still retaining their tropical characteristics such as high moisture content and strong winds. Two events with significant impacts on the coastal NEUS in the last two decades include Hurricane Irene in 2011 and Hurricane Sandy in 2012. Both systems caused severe flooding in parts of the coastal NEUS. Extreme precipitation events in the NEUS also occur from the remnants of tropical storms. Significant flooding and damage in New York City in late September 2023 caused by the remnants of tropical storm Ophelia and flooding in New Jersey in 2021 caused by the remnants of Hurricane Ida are notable examples of remnants of tropical systems affecting the NEUS.

Winter storms: A projected increase in winter precipitation in the future will result in more rainfall and less snowfall due to atmospheric warming. This leads to model projections
indicating a decrease in snowfall frequency in the NEUS in the future (Zarzycki, 2018). This, however, does not discount the likelihood of individual high-impact snowfall events. In fact, during sufficiently cold conditions, snowstorms in the future can drop more snow because of the ability of the warmer atmosphere to hold more moisture (Zarzycki, 2018). In fact, the frequency of heavy snowfall in the NEUS has increased over the past three decades (Whitehead et al., 2023), which is likely caused by interactions between warming in the western Atlantic Ocean and frequent Arctic air outbreaks (Cohen et al., 2018). Higher winter warming will result in an overall increase in days and nights with temperatures above freezing, leading to decreases in snow cover as well as snow depth (Burkowski et al., 2022). These changes are likely to impact coastal and southern regions of the NEUS first with high elevation regions remaining more resilient (Burkowski et al., 2022). A projected increase in winter precipitation in the form of rainfall is also projected to increase surface runoff and peak river flows in winter in the future (Siddique and Palmer, 2021; Siddique et al., 2020).

**E) SEA LEVEL RISE**

The NEUS has been experiencing a steady rise in sea level over the last century (Fig. 1.9). This increase of about 1 ft (12 inches) since 1900 is higher than the rise in global sea level (Sweet et al., 2022). As of 2018, the global mean sea level rose by about 0.5-0.8 feet relative to 1900, and 0.2-0.5 feet relative to 1971. The IPCC AR6 Report (Fox-Kemper et al., 2021) concludes that it is *virtually certain* that the global mean sea level will continue to increase this century and is projected to rise 0.60-0.75 feet by 2050 and 1.25-2.5 feet by 2100, relative to 1971. Local changes in sea level are determined by a complex combination of several geological, oceanographic, and atmospheric factors that operate on different timescales. On long timescales (decades and longer), the main driver of sea level is changes in the volume of water in the ocean. The volume can increase due to the thermal expansion of the water as it gets warmer and due to the addition of water from melting land ice (ice sheets in Greenland and Antarctica, glaciers around the world). Sea level in the coastal NEUS also varies substantially from year to year due to variations in atmosphere and ocean circulations. For instance, high sea levels along the NEUS coast in 2009 and 2010 have been linked to changes in the ocean circulation in the Gulf Stream region and changes in wind circulation associated with basin-wide variations in the atmospheric...
variability pattern in the North Atlantic basin (Goddard et al., 2015; Domingues et al., 2018; Piecuch et al., 2019).

Figure 1.9: (a) Relative sea level trends along the Northeast U.S. coast, with arrows representing the direction and magnitude of change. (b) Linear trend in relative sea level in Portland, ME. (c) Linear trend in relative sea level in Atlantic City, NJ. Values in the top right in
panels b and c indicate the rate of sea level rise in millimeters per year (mm/yr) (Source: NOAA Tides & Currents https://tidesandcurrents.noaa.gov/sltrends/sltrends.html)

Rising sea levels can enhance the impact of storm surges during winter storms (e.g., Nor’easters), hurricanes, and other severe weather events. Higher sea levels mean that storm surges can penetrate further inland, causing more extensive flooding and damage. The rise in sea level is also contributing to an increase in the frequency and duration of minor coastal flooding events (called ‘nuisance’ or ‘sunny day’ flooding) along the U.S. east coast (Sweet et al., 2018; Ezer, 2020). Additionally, storm surges and coastal flooding can have significant impacts on groundwater in coastal areas. There is mounting evidence that the intrusion of saltwater has been contaminating freshwater resources (Panthi et al., 2022; USDA, 2020) in the NEUS and around the US, making them unsuitable for drinking and agricultural use. Higher sea levels lead to increased coastal erosion as waves reach further inland, wearing down shorelines and threatening coastal infrastructure.

F) DROUGHTS

Droughts do not feature prominently in studies focusing on the present-day and future climate of the NEUS since the region receives abundant precipitation throughout the year and has been experiencing an increase in mean and heavy precipitation in recent decades as described earlier. The Palmer Drought Severity Index (PDSI) is an effective index to determine droughts since it takes into account the effect of evaporative demand—associated with temperature—on moisture balance rather than relying on precipitation alone. Positive PDSI values indicate wet conditions while negative PDSI values indicate droughts. A closer look at PDSI anomalies over the twentieth century suggests several instances of drought conditions in the region (Fig. 1.10). Figure 1.10 shows the monthly PDSI time series for the Northeast between 1901 to 2005. It clearly shows at least three droughts, in the 1930s, 1960s, and early 2000s, that are of different duration but similar magnitudes.
Figure 1.10: The monthly mean Palmer Drought Severity Index (PDSI) in the Northeast U.S. between 1895 and 2022. The green bars (positive PDSI values) indicate wet conditions while the orange bars (negative PDSI values) indicate drought conditions. The blue line shows the trend over the entire period. (Source: NOAA Climate at a Glance)

The simplest and most widely accepted view for the occurrence of droughts in a warming world is related to an increase in evaporative demand (i.e., an increase in evaporation and transpiration) with warming (Cook et al., 2014; Dai, 2013). There is indeed an observed increase in soil moisture drought conditions (agricultural droughts) over the last 50 years in many parts of the U.S. even in the absence of significant precipitation deficits (i.e., meteorological droughts; see Mazdiyasni and AghaKouchak, 2015, Cook et al., 2014), occurring mainly when low precipitation and high temperature extremes occur more or less concurrently.
The maximum number of consecutive dry days (CDD) is a useful metric to indicate the persistence of dry conditions. The metric used here counts the number of days with daily precipitation below 1mm. We use this metric to examine the current and future changes in dry conditions. We find that there are years with a small increase in the maximum number of consecutive dry days toward the end of this century (Fig. 1.11). This suggests that an overall increase in precipitation in the future will not necessarily result in persistent wetness. The increase in the number of consecutive dry days combined with increased temperatures can, in fact, trigger and intensify short-term drought conditions in the region. It is worth noting that moderately abnormal temperature and precipitation conditions can combine to create extreme conditions in water availability. For instance, a heavy precipitation event following drought conditions may lead to flooding, waterlogging, and erosion because the drought can result in compact soils reducing their ability to absorb water. In such cases, plants may continue to experience water stress despite heavy rains. In the case of intense storms with excessive runoff, the water may infiltrate inadequately and not reach the root zones of plants.
IV. SUMMARY

This chapter summarizes the observed and projected changes in climate for the Northeast U.S. The NEUS has warmed by 1.4°C (2.5°F) since 1895 and has also experienced a steady increase in total annual precipitation over that period. The pattern of warming over the historical period has been spatially heterogeneous with the coastal regions warming more than the interior parts. Climate models indicate that average and extreme temperatures are projected to increase but the exact magnitude of the increases is dependent on the future trajectory of greenhouse gas emissions. Average and extreme precipitation are also projected to increase this century, but year-to-year projections remain highly variable. Due to high variability, precipitation projections for different emissions scenarios overlap completely throughout this century, indicating more attention should be paid to precipitation variability than any particular emissions scenario. Despite the projections indicating overall wet conditions in the future, the region may also experience seasonal droughts caused by limited precipitation and high temperatures within brief timeframe. Climate projections reported here are consistent with those reported previously, providing reliable trajectories for risk assessment and management decision-making.

The climate change overview presented in this chapter provides context for subsequent chapters. Chapter 2 delves into biological responses, exploring physiological and thermal threshold-based responses to factors like temperature, sea level rise, and extreme storms discussed here. Chapter 3 centers on Climate Change Vulnerability Assessments (CCVAs), which characterize risk through the exposure component of vulnerability. The climate data presented in this chapter can serve as a valuable resource for characterizing climate change in new CCVAs across the region. In addition, the adaptation frameworks summarized in Chapter 4 can use regional climate data from this chapter to characterize the amount of change and overall variability expected when choosing approaches and strategies to protect habitats, plan corridors of movement, and identify regions across the landscape where certain actions are more likely to succeed (e.g., areas with lower warming or fewer extreme events) compared to others (e.g., areas experiencing enhanced warming or sea level rise).
V. LITERATURE CITED


CHAPTER 2: BIOLOGICAL RESPONSES TO CLIMATE IMPACTS OF THE NORTHEAST REGIONAL SPECIES OF GREATEST CONSERVATION NEED

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†Current address

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   B) ESTIMATING RANGE SHIFTS .................................................................................. 8
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Key findings

- The most climate data-poor taxonomic groups included most invertebrate groups, including fireflies, dragonflies, and mayflies, as well as vertebrate groups such as salamanders and small mammals.

- The most data-rich taxonomic groups included marine invertebrates, birds, diadromous and marine fishes; however, a few species had a disproportionate number of studies compared to others within their taxa groups including American Lobster (*Homurus americanus*), Atlantic Cod (*Gadus morhua*), Monarch (*Danaus perplixus*), and Alewife (*Alosa pseudoharengus*).

- Discrepancies in taxonomic and biological response research across taxonomic groups and species highlight areas for new research to fill basic knowledge gaps and prioritized *in situ* monitoring to detect and track populations in areas of high exposure and projected climate change as identified in Chapter 1.

- The document is organized based on taxa groups, with the most information first and the least information last. State experts can use adaptation case studies, frameworks, and tools summarized in Chapter 4 to develop species-specific actions to develop new monitoring and research plans and reduce uncertainty.

I. INTRODUCTION

A) OVERVIEW

There are three general types of species’ responses to climate change: move, adapt, or go extinct. While reality is undoubtedly more complex, many species will respond in these general ways over the coming decades and millennia. Recent research suggests that many taxa are already shifting higher in elevation and latitude, but results are inconsistent (Rubenstein et al., 2023), and marine taxa appear to track temperature shifts more closely than terrestrial species (Lenoir et al., 2020). Some level of adaptation to a changing climate is likely for many species; however, species are more quickly shifting the timing of important parts of their life history (phenology) than they are their morphology or physiology; it remains to be seen if these shifts
will be substantial enough to shield many species from serious population declines or extinction (Radchuk et al., 2019). Over the past century, extinction rates appear higher than speciation rates (Dornelas et al., 2023), and future projections estimate that as many as 14-32% of all species may go extinct by ~2085 (Wiens and Zelinka, 2024). Given the immensity of conserving rapidly declining species facing a myriad of threats, including habitat loss, invasive species, and pollution, adding climate change threats to the equation is yet another complexity that needs to be anticipated and mitigated (Armsworth et al., 2015).

The added difficulty of including climate change in conservation planning (see Chapters 3, 4, and 5) is challenging for decision-makers. Two of the most difficult aspects of addressing climate change in conserving species are: 1) anticipating how climate will change in the long- and short-term future depending on different carbon emission scenarios (See Chapter 1) and 2) the uncertainty of how target species may directly respond to changing temperature and precipitation patterns, as well as indirect abiotic and biotic changes, which are covered in this chapter. To help address this, ten years ago, the Northeast Climate Adaptation Science Center (NE CASC) provided a report to the northeastern and midwestern U.S. states (Staudinger et al., 2015) to document the extent of the climate change research on the Regional Species of Greatest Conservation Need (RSGCN) and facilitate the incorporation of that information into their 2015 State Wildlife Action Plans (SWAP) revisions. This chapter is an update to the information provided in Chapter 3 of the 2015 report (Staudinger et al., 2015) and reports on newly available information for the updated RSGCN list of 418 species (revised from 367 species in 2015; see Appendix 2.1) in the footprint of the Northeast Association of Fish and Wildlife Agencies (NEAFWA) region.

Projecting how species and populations will respond to climate change is complex since many dimensions of change are involved. A species may shift its range across latitude, elevation, or water depth to stay within its physiological tolerances of temperature, precipitation, or ocean acidification levels. In doing so, they may encounter various obstacles, such as altered interactions with prey or competitors (e.g., Carlson et al., 2013; Sandor et al., 2021). A species may also shift in its phenology (i.e., the timing of life events), such as the timing of breeding and migration, to take advantage of changing resource availability. For example, birds have started breeding ~2-3 days earlier per decade since the early 19th century (Romano et al., 2022). However, they may not be able to shift in the same way as the resources that species need to feed
their offspring or provide enough fuel to finish their migration successfully. Some species may migrate shorter distances and, ultimately, set up year-round resident populations in areas they previously only visited during migration due to changes in resource availability (Visser et al., 2009). Some species may have much broader physiological tolerances or ability to acclimate than are apparent since other processes may be limiting their current distribution (e.g., competition), which affects the ability of current models to accurately project where they may be able to persist in the future given various climate change scenarios (MacPherson et al., 2021, Perret et al., 2023). Moreover, some species may be able to rapidly adapt to changing climate conditions through changes in their physiology or morphology, even on the scale of a few generations (e.g., Grant and Grant, 1993), which further complicates our understanding of how species may respond to changing climates, communities, and resource availability. Research for each of these three main themes: shifting range, shifting phenology, or changes in morphology/physiology are highlighted for each RSGCN in this report.

Many long-term, population-level shifts and changes will result from demographic processes. For instance, a poleward range shift of a species tracking cooler temperatures and precipitation patterns may not necessarily result from individuals moving further poleward, especially if individuals of a species cannot disperse great distances. Rather, it would be more likely the result of an iterative process of higher fitness in the cooler parts of the range and lower fitness in the warmer parts of their range, leading to population growth in the cooler parts of the range and population decline and ultimately, extirpation, in the warmer parts of the range. Some evolutionary processes, like gene flow among sub-populations, could be maladaptive or limit local adaptation to rapidly changing climate or, in some cases, have the opposite effect (Botranger and Angert, 2019). For example, for highly mobile species, such as birds, individuals adapted to the cooler “core” part of their range may continue to breed with individuals at the warmest edge, effectively limiting natural selection of that subpopulation’s ability to adapt to the changing conditions. These complexities, among others, further highlight the challenges of planning and effectively managing Northeast RSGCN populations in the face of climate change.

Given these uncertainties, learning more about how species have been responding will go a long way to help create more accurate models that better reflect what will happen in the future and lead to more effective conservation plans. It is likely that in the next 50-80 years, novel species may enter the Northeast region, while others may shift out; further, species that were
previously rare in the Northeast may redistribute with the majority of their range occurring in the region in the coming years (e.g., the Indiana Bat; Loeb and Winters, 2013), and require an examination of both current and future habitat needs (Zhu et al., 2022; Sales and Parrott, 2024). Over the last 10 years, much has been learned about how some RSGCN have responded to climate change; however, there is a major discrepancy in climate change research among taxa, where marine species have been studied disproportionately more than terrestrial species in recent years.

II. METHODS
A) LITERATURE SEARCH

For this chapter, three systematic literature searches were conducted using the PRISMA 2020 framework for repeatable systematic reviews (Page et al., 2021) in the Web of Science. To generate keywords, the co-authors of this report were surveyed for climate change-related suggestions. Additional keywords were identified using the R package “LitSearchR” (Grames et al., 2019) by searching the Web of Science using a set of “naive” search terms (“temperature,” “drought,” “acidification,” “phenology,” “precipitation,” and “adaptation”). The first 2,000 papers identified by the search were downloaded and entered into LitSearchR to remove duplicates, create a keyword co-occurrence network of the remaining articles, and identify top keywords (Grames et al., 2019). Ten keywords were identified through LitSearchR and 26 from co-author suggestions for 36 keywords (Table 2.1).

Web of Science was searched in April 2022 to capture peer-reviewed journal articles published from 2013 to 2022 about the RSGCN in the northeastern U.S. since the last NE CASC regional SWAP climate change synthesis (Staudinger et al., 2015). Keywords listed in Table 2.1 were searched in combination with the scientific and common names of all 382 RSGCN (see Appendix 2.1). A second search, also limited from 2013 to 2022, was conducted in October 2022 using the same keywords in combination with the scientific and common names of the 36 Proposed RSGCN (Appendix 2.1). A third and final search was conducted in April 2023 that included all keywords plus RSGCN and Proposed RSGCN names limited to 2022 to 2023 to capture any recent research published since the first two searches. In all, the three searches yielded a total of 7,766 scientific articles.
Once the literature searches were completed, the information for each article was downloaded into a spreadsheet. First, duplicate studies (N = 20) were removed. The titles and abstracts were then reviewed to verify that the article was, in fact, about one of the RSGCN or Proposed RSGCN species, if the article directly addressed a biological response to climate change, and if the research in the article took place in the northeastern U.S., eastern Canada, the northwestern Atlantic Ocean, or was experimental. In some cases, the full article was reviewed if the abstract and title lacked the necessary information to determine relevance. A total of 6,917 articles were excluded from further consideration as they did not meet the search criteria conditions as belonging to an RSGCN population occurring within the geography of the Northeast region. The remaining 829 articles were then further reviewed for inclusion in this report. Of those 829 articles that were read in their entirety, 280 are cited in this report (see Figure 2.1) and are supplemented by other relevant research articles found opportunistically or identified by co-authors or reviewers during the development of the chapter content. Throughout the rest of this document, the number of papers reported are the total number of papers found through the literature search and those found opportunistically to reflect the actual number of research papers containing relevant information.

Table 2.1. Climate change-related keywords used in each of the three literature searches.

<table>
<thead>
<tr>
<th>Keyword</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>acidification, adaptation, climate &quot;novel assemblages&quot;, climate &quot;species interactions&quot;, climate adaptive capacity, climate biological response, climate distribution shift/change, climate ecological response, climate extreme event, climate flooding, climate hurricanes, climate impact, climate mismatch, climate nor’easters, climate plasticity, climate regime shift, climate sensitivity, climate storms, climate trophic, drought, global change, heatwave, phenology, precipitation, range shift/change/expansion/contraction, temperature</td>
<td>Co-author suggestion</td>
</tr>
<tr>
<td>climate change, climate change impact, environmental change, extinction risk, global warming, higher elevation, sea-level rise, species distribution, suitable habitat, water availability</td>
<td>LitSearchR</td>
</tr>
</tbody>
</table>
Figure 2.1. Screening process for the systematic literature review results in the PRISMA 2020 Framework.

The content of this chapter is organized to reflect the weight of evidence across RSGCN. Taxonomic groups that appear first in the document have the greatest number of literature results and corresponding articles, while groups appearing later in the document have few to no research articles found on their responses to climate change (Table 2.2).

Table 2.2. Number of Scientific Articles Cited in this Chapter by Taxonomic Group.

<table>
<thead>
<tr>
<th>Taxonomic Group</th>
<th>Number of Articles Cited</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marine Invertebrates</td>
<td>107</td>
</tr>
<tr>
<td>Birds</td>
<td>90</td>
</tr>
<tr>
<td>Diadromous Fishes</td>
<td>66</td>
</tr>
<tr>
<td>Marine and Coastal Fishes</td>
<td>57</td>
</tr>
<tr>
<td>Marine Turtles</td>
<td>44</td>
</tr>
<tr>
<td>Butterflies and Skippers</td>
<td>27</td>
</tr>
<tr>
<td>Marine Mammals</td>
<td>15</td>
</tr>
<tr>
<td>----------------------------</td>
<td>----</td>
</tr>
<tr>
<td>Carnivores and Small Mammals</td>
<td>14</td>
</tr>
<tr>
<td>Bats</td>
<td>11</td>
</tr>
<tr>
<td>Freshwater Turtles</td>
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<tr>
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<td>5</td>
</tr>
<tr>
<td>Bees</td>
<td>3</td>
</tr>
<tr>
<td>Snakes</td>
<td>2</td>
</tr>
</tbody>
</table>

B. ESTIMATING RANGE SHIFTS

Long-term species occurrence data from bottom trawl surveys operated by NOAA Fisheries provided a unique opportunity to present comparable metrics over equivalent spatial and temporal scales. The Distribution and Mapping Analysis Portal (DisMAP) was used to extract observed geographical range and depth shifts along the U.S. Atlantic coast for the majority of RSGCN marine (N=13) and diadromous fishes (N=3), elasmobranchs (e.g., skates, N=3), and macroinvertebrates (N=3). Data for most species spanned the Northeast continental shelf region from the Gulf of Maine to the Mid-Atlantic Bight (Maine – Cape Hatteras, North Carolina); however, data for at least one species (e.g., Weakfish [*Cynoscion regalis]*) came from the Southeast region (North Carolina to Florida) where it has historically been more abundant. Range shift metrics presented in the text have comparable maps available online for those interested in analyzing the data further (NOAA Fisheries, 2022).
III. SPECIES RESPONSES

A) MARINE INVERTEBRATES

Table 2.3. RSGCN marine invertebrate species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
</thead>
<tbody>
<tr>
<td>American Lobster</td>
<td><em>Homarus americanus</em></td>
<td>56</td>
</tr>
<tr>
<td>Atlantic Sea Scallop</td>
<td><em>Placopecten magellanicus</em></td>
<td>10</td>
</tr>
<tr>
<td>Bay Scallop</td>
<td><em>Argopecten irradians</em></td>
<td>24</td>
</tr>
<tr>
<td>Horseshoe Crab</td>
<td><em>Limulus polyphemus</em></td>
<td>17</td>
</tr>
</tbody>
</table>

Figure 2.2. Number of scientific articles for RSGCN marine invertebrates by biological response type.
American Lobster

Overview

American Lobster uses different habitats, each influenced by variable environmental conditions, depending on its life stage. The larvae are pelagic and transition between three planktonic instars (Phillips et al., 2013). During the post-larval instar phase, American Lobster transition from being pelagic at the water’s surface, subject to transport by wind, to settling on the benthic seafloor. Post-larvae can swim and move vertically and horizontally in the water column, capable of diving to select a benthic habitat for settling that provides shelter. Typically, this transition to benthic habitat occurs in shallower water above the summer thermocline. After they settle, they transition to the juvenile life stage, where they exhibit cryptic behavior strongly associated with rocky shelters. As very young juveniles, they are thought to be suspension feeders. Their behavior and habitat use transitions again as the juveniles grow into adults, with increasing use of habitats farther from shelters (Phillips et al., 2013). They also seasonally migrate between inshore and offshore waters (Phillips et al., 2013; Goldstein and Watson, 2015). Adult lobsters molt every one to two years. This shifting life history and habitat use exposes American Lobster to variable climate impacts depending on the depth and temperature associated with each life stage (Quinn et al., 2013; Goldstein and Watson, 2015; Quinn and Rochette, 2015; Barret et al., 2017; Goode et al., 2019).

Juvenile and adult American Lobsters are positioned in the middle of the marine food web, feeding primarily on benthic invertebrates but also plant material and potentially fish, while being important prey for marine demersal and groundfish like Atlantic cod and other marine fishes and crustaceans. With declining groundfish populations in the Gulf of Maine, predation pressure on American Lobster has decreased and is thought to be a contributing factor alongside warming temperatures for increasing lobster abundance in the northern portion of its range (Phillips et al., 2013; Steneck and Wahle, 2013).

Shifts in Range, Elevation, or Depth

American Lobster ranges from North Carolina north to Labrador, Canada, in shallow coastal waters in the northern portion of its range and deeper offshore waters in the southern portion of its range (Phillips et al., 2013). In Atlantic Canada the species is primarily found in waters less than 50 meters deep. American Lobsters are distributed widely from the coast to
submarine canyons at the continental shelf break in the middle of its range. From Long Island Sound south through the Mid-Atlantic Bight, they are increasingly distributed in colder, deeper waters at the edge of the continental shelf (Phillips et al., 2013). As water temperatures increase with climate change, the southern end of the species range may likely become less metabolically favorable, while cooler, more northern waters will be more hospitable during summer months (Phillips et al., 2013). Casey et al. (2023) found that adult lobsters in southern New England have shifted offshore into deeper waters to release larvae in response to rising water temperatures in nearshore areas.

Over the past several decades, American Lobster populations have shifted northward, with increasing abundance in the Gulf of Maine and declining and disappearing populations in southern New England (Phillips et al., 2013; Wahle et al., 2015; Goode et al., 2019; Oppenheim et al., 2019; Mazur et al., 2020; NOAA Fisheries, 2022; Casey et al., 2023). In Narragansett Bay, Rhode Island, summer lobster nurseries contracted from the middle portions of the bay between 1990 and 2011-2012 to outer portions that are deeper, coincident with an increase in summer surface temperatures of about 2°C (Wahle et al., 2015). Goode et al. (2019) suggested that the northward increase in lobsters results from complicated interactions between climate change, local oceanographic conditions, and the settlement behavior of lobster larvae.

Between 1974-1977 to 2019-2022, the spring range of American Lobster shifted 1.68 degrees (187.04 km) north and contracted by 2.58 degrees (285.34 km). During the fall season, American Lobster moved 1.54 degrees (171.08 km) north and contracted its range by 1.93 degrees (213.77 km) from 1974-1976 to 2019-2022. The spring distribution of American Lobster shifted 11.3 meters shallower during spring and 21.1 meters deeper during fall over the same periods (NOAA Fisheries, 2022).

Mazur et al. (2020) modeled the distribution of suitable habitats for American Lobsters in the Northeast, which is strongly influenced by environmental conditions. They found that abundance and distribution were driven by temperature, bathymetry characteristics, and secondary productivity of zooplankton. Given these drivers, the availability of suitable habitat has increased in the Gulf of Maine and decreased in southern New England. Evidence also indicates a decline of inshore habitat suitability in the Gulf of Maine and an increase offshore in deeper, cooler waters (Mazur et al., 2020). In Nova Scotia, Greenan et al. (2019) forecasted that the distribution of American Lobsters is not likely to change or improve.
Tanaka et al. (2020) projected future range shifts of American Lobster and Atlantic Sea Scallops in the Northeast with shifts in environmental conditions due to climate change. Their results projected that American Lobsters will move further offshore over the next 80 years with anticipated changes to bottom temperature and salinity. Tanaka and Chen (2016) similarly projected increasing inshore habitat suitability for juvenile and adult lobsters during the spring (April to June) but no trend for the fall (September to November).

American Lobsters were projected to experience a significant loss of relative biomass during spring and fall by 2050 across the Northeast continental shelf, with slightly larger declines during fall in the Southern New England – Mid Atlantic compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). Behan et al. (2021 & 2022) also modeled the distribution of American Lobster under the RCP 8.5 scenario, finding that forecasted habitat suitability in the Gulf of Maine for 2028-2055 varies with the temperature and salinity of bottom water, sediment grain size, the distance offshore, and latitude. Forecast results depended on the spatial scale and assumptions about the models' environmental parameters, generating uncertainty (Behan et al., 2021 & 2022; Hodgdon et al., 2021 & 2022; Quinn et al., 2022).

**Shifts in Phenology**

Water temperature is a key driver of American Lobster post-larval settlement and population dynamics (Annis et al., 2013; Barret et al., 2017). Annis et al. (2013) found that settlement patterns of post-larval lobsters may be influenced by small variations in water temperature, with “either behavioral avoidance of colder settlement sites or elevated post-settlement mortality of post-larvae settling at colder sites”, resulting in a potential disconnect between the abundance and settlement of post-larvae in warmer inshore areas and cooler offshore areas.

Haarr et al. (2020) documented an advancement in the timing of egg hatching in the southern Gulf of St. Lawrence by five weeks from 1989 to 2014 in response to warming temperatures. The rate of clutch development increased by 40% in the spring. Their results indicated that the earlier hatching time is a response to environmental conditions the females were exposed to 6 to 18 months earlier. The earlier hatch timing could result in a phenological mismatch with prey species for lobster larvae (Haarr et al., 2020).
Monitoring studies by Staples et al. (2019) in three regions of the Gulf of Maine found different patterns in the timing and suddenness of initial molts between regions, stages of maturity, and sexes. Molting was variable with molts that were earlier associated with warmer temperatures, especially in inshore areas, but this response was not uniform and challenging to quantify as a molt time series connected with bottom temperatures (Staples et al., 2019).

Some egg-bearing female lobsters make seasonal migrations between inshore and offshore areas, exposing their eggs to thermal conditions that differ from egg-bearing females who do not migrate (Goldstein and Watson, 2015). Goldstein and Watson (2015) found that the egg development rate did not change with exposure to different fall temperatures, but exposure to different spring temperatures led to faster development of eggs in warmer inshore areas. Eggs from inshore areas hatched approximately 30 days sooner than those in offshore areas and had significantly shorter development times from the onset of an eyespot to their hatch. Their results suggest that the seasonal movements of egg-bearing female lobsters influence the location and timing of egg hatching, with implications for larval transport and recruitment between inshore and offshore areas (Goldstein and Watson, 2015).

Warming water temperatures at the southern end of the range of American Lobster are associated with an increase in seasonal outbreaks of epizootic shell disease (Groner et al., 2018). Groner et al. (2018) determined that a phenological mismatch is linked to disease outbreaks in Long Island Sound. Elevated water temperature in the spring correlates with earlier lobster molting in the spring, increasing the intermolt period during the summer when disease prevalence peaks in the fall. Warmer summer water temperatures increase the rate of new infections, with an 80% infection rate in warmer summers and a 30% infection rate in cooler summers. Lobsters that are infected had a mortality rate of greater than 50%. Groner et al. (2018) suggest that the phenological mismatch between molting and disease outbreaks due to rising temperatures will likely lead to future population-level impacts as waters continue to warm.

Changes to Morphology or Physiology

American Lobster larvae and post-larvae have been documented to have negative physiological responses to increasing water temperatures and acidification (Benestan et al., 2016; Hartington and Hamlin, 2019; Harrington et al., 2020a & 2020b; Lopez-Anido et al., 2021; Niemisto et al., 2021; Powell et al., 2023). McMahan et al., (2016) found that juvenile lobsters in
Maine are growing faster in warmer years and molting more frequently. Waller et al. (2019) correlated a reduction in female size of maturity in Maine of up to 50% with increasing water temperature. Haarr et al. (2018) documented a reduction of up to 30% in the size of maturity of female lobsters in Canada over the past 10-80 years but did not find a correlation with increasing temperatures but rather with increasing fishing pressure. Goode et al. (2019) determined that the thermal tolerance of lobster post-larvae is between 12℃ and 20℃ and that surface waters that are thermally stratified in the summer in the southwestern Gulf of Maine have historically been within this thermal envelope. Surface waters in the Gulf of Maine are warming faster than bottom waters, however, creating a steep temperature gradient with depth. They conclude that the potential expansion of thermally suitable areas for settlement by lobster larvae is inhibited by the thermal stratification of bottom waters in the southwestern Gulf of Maine. In the northeastern Gulf of Maine, tidal mixing prevents thermal stratification, and the favorable area for larval settlement has expanded more (Goode et al., 2019). Casey et al. (2023) found that increasing water temperatures in the nearshore areas of southern New England has significantly increased thermal stress on recently settled post-larvae.

Quinn (2017) investigated the thermal stress and tolerance levels of lobster larvae and documented sublethal thermal stress with exposures to water temperatures of 20-26℃ with potential lethal effects with longer exposures. With projected exposure to surface water temperatures over 30℃ with climate change, there could be negative population-level effects (Quinn 2017). Quinn et al. (2013) found that warmer temperatures caused faster larval development and shorter durations of larval stages, and colder temperatures resulted in 38% shorter development times in the northern Gulf of St. Lawrence. Larvae from this colder region had longer development times at warmer temperatures than previous studies have documented for larvae from warmer regions, indicating regional variability in the functional relationship between larval development time and temperature (Quinn et al., 2013).

Harrington et al. (2019) found physiological trade-offs in post-larval lobsters with warming temperatures, with warming temperatures increasing growth rates but with physiological stress and loss of genetic diversity. These trade-offs could result in reduced adaptive capacity for climate change. Nielsen and McGaw (2016) determined that juvenile lobsters avoid temperatures lower than 8℃ and higher than 20℃, with a mean temperature
preference of about 16.2°C. Their thermal preferences were not affected by size, origin (wild versus laboratory-raised), or prior acclimation (Nielsen and McGaw, 2016).

When juvenile lobsters were offered a choice between food, shelter, or temperature, they consistently chose the environment with a shelter over those with food, even if the temperature was thermally unfavorable - suggesting juvenile lobsters make behavioral trade-offs to maintain optimal fitness and survival (Nielsen and McGaw, 2016). Wang et al. (2016) found that American Lobsters eat more food with increasing temperature and that they process their meals through their digestive system faster with increasing temperature. The results suggest that feeding cues are driven by stomach emptying regardless of temperature (Wang et al., 2016).

Experiments conducted by Noisette et al. (2021) determined that lobster larvae have a higher tolerance to acidification, with no significant changes in development, morphology, mineralization, or survival time, but juvenile lobsters had the opposite pattern, with an increased time of development and reduced survival with acidification. They found that tolerance for acidification decreased until larvae metamorphosed into juveniles, after which their sensitivity increased (Noisette et al., 2021). Menu-Courey et al. (2019) found an increase in juvenile lobster mortality, an increase in aerobic capacity, and slower development with increasing acidification. McLean et al. (2018) found reduced growth and an increased vulnerability of juvenile lobsters to acidification. Acidification may negatively affect the calcification processes of lobsters during molting, with potential implications for mortality (Nagle et al., 2018).

Waller et al. (2017) assessed the interactive effects of acidification with temperature on American Lobster, showing that projected warming by the end of the century will have a larger negative effect on larval survival than acidification but that acidification may affect larval behavior and metabolism in complex ways. Klymasz-Swartz et al. (2019) found that acidification negatively impacts every life stage of American Lobster and projected that by 2300, they likely would not be physiologically capable of acclimating to projected acidification levels.

Population Changes

The size and age of maturity of American Lobster varies regionally across its range, with females in the southern and warmer portion of the range (i.e., southern New England) maturing three to four years faster than those in the northern portion of the range (i.e., Bay of Fundy; Phillips et al., 2013). Lobsters in warmer waters have been found to mature at smaller sizes (Le
Bris et al., 2017; Haarr et al., 2018). Khalsa et al. (2023) found that the size at maturity was an important driver of lobster productivity, suggesting that changes to lobster growth and maturity due to climate change may positively impact the population in the Gulf of Maine.

American Lobster populations have skewed sex ratios, with different populations either male- or female-dominated (Jury et al., 2019). The skewed sex ratios are thought to be a result of both seasonal migrations inshore and offshore and selective harvest regulations and sex differences in catchability. Changing environmental conditions with climate change could impact lobster reproduction if local sex-specific populations shift as a result (Jury et al., 2019).

Local populations of American Lobster post-larvae do not appear to be thermally adapted, with no significant differences in settlement behavior with the geographic origin of their female parents (Barret et al., 2017). Temperature was found to be a significant driver for larvae's survival and settlement behavior and the energetic conditions of post-larvae, suggesting that there are connections between development temperature and survival and settlement (Barret et al., 2017). Jaini et al. (2018) found that these connections with temperature and larval survival and settlement vary regionally, with positive associations for settlement with sea surface temperature in southern New England, Georges Bank, and southern Nova Scotia during the summer months. Settlement was only associated with sea surface temperature in the Gulf of Maine in the vicinity of the settlement site, and settlement in the Bay of Fundy had no correlation with sea surface temperature. Their results were consistent with residual oceanic flow driving larval distribution and settlement (Jaini et al., 2018).

American Lobster populations are controlled by intense predation by large finfish, particularly Atlantic Cod. Steneck and Wahle (2013) argued that American Lobsters are evolutionarily adapted to predation pressures with an extended brood period, large larval size, and settlement habitat selection for shelters. Removing coastal predators from the Northwest Atlantic has allowed more population growth since 1980 in the US and Canada (Steneck and Wahle, 2013). Le Bris et al. (2018) modeled the synergistic effects of climate change, predation, and harvest on American Lobster population productivity. Their results found that climate change amplifies harvest impacts and vice versa, with the species’ reproductive potential driven by both (Le Bris et al., 2018).

As population densities increase with warming waters, the vulnerability of American Lobsters to disease also increases (Steneck and Wahle, 2013; Groner et al., 2018). The
prevalence of epizootic shell disease is increasing along the coast of Maine. Molting lessens the severity of the disease or eliminates it. Experiments conducted by Groner et al. (2018) found that diseased lobsters were more likely to molt and die and that those trends did not vary with temperature, and the temperature regime did appear to impact a physiological mechanism that could mitigate the effects of epizootic shell disease.

Laufer et al. (2013) investigated a major die-off of the Long Island Sound population of American Lobster in 1999, finding that the causes of mortality were the cumulative effects of climate change stressors (temperature, acidification, and dissolved oxygen levels), shell disease, and pollution with endocrine-disrupting alkylphenol chemicals. The chemical pollutants negatively impacted the survival of larvae, molting and hardening of shells, and interfered with the metamorphosis between larvae and juvenile life stages (Laufer et al., 2013).

A meta-analysis by Boudreau et al. (2015) found that, over the entire range of American Lobster, there is an interaction between predation and temperature, with predation the dominant population driver at the warm and cold extremes but not in the center of the range. Temperatures positively affected the recruitment of Lobsters at the warm range extremes. Lobster abundance did not follow the fishing harvest, but fishing effort followed Lobster abundance over time. Their results suggested that lobster populations will intensify at the thermal range boundaries of the species and weaken in the central core of the range (Boudreau et al., 2015).

Oppenheim et al. (2019) enhanced population models for American Lobster to account for local bottom temperature and disease prevalence. Their results project that Gulf of Maine harvest populations will decline to near historical levels in the next decade and that the southern New England populations will not recover (Oppenheim et al., 2019).

Tanaka et al. (2019) found that bottom temperature and salinity are drivers of lobster abundance during the spring and projected increasing populations in the Gulf of Maine. Tai et al. (2021) investigated the population effects of ocean acidification on American Lobster, finding that juveniles are the most vulnerable to negative impacts, but all life stages are impacted to some degree. Their population model indicated that the increasing magnitude of climate change effects would outweigh any population gains from harvest management (Tai et al., 2021).
Indirect Effects

The warming waters, thought to be the primary driver of the decline of juvenile American Lobsters from shallower inshore habitats in southern New England, may be amplified by the arrival and spread of invasive predatory and competitor species like the Asian Shore Crab (*Hemigrapsus sanguineus*) since the 1990s (Wahle et al., 2015).

Overall, the American Lobster is a well-studied species, but many studies have focused on information needs with fisheries management or socioeconomic implications. Recent literature does not address how changing ocean circulation patterns and strengths may impact larval distribution and settlement, particularly any interactive effects with warming temperatures. Needed research to fill data gaps for American Lobster includes additional monitoring of larval settlement and juvenile nursery habitat in deep water in southern New England (Wahle et al., 2015), development of molt time series, and quantifying the connection to bottom temperature (Staples et al., 2017), quantification of the rate of larval development in nature and how it may vary over time and space (Quinn et al., 2022).

Horseshoe Crab

*Shifts in Range, Elevation, or Depth*

Between 1974-1977 to 2019-2022, the spring range of Horseshoe Crabs shifted 0.68 degrees (76.04 km) north and contracted by 0 degrees (0.13 km). During the fall season, Horseshoe Crabs moved 93 degrees (103.89 km) north and contracted its range by 1.98 degrees (219.97 km) from 1974-1976 to 2019-2022. The spring distribution of Horseshoe Crabs shifted 9.6 meters shallower during the fall over the same periods (NOAA Fisheries, 2022).

Horseshoe Crabs are projected to experience a significant gain of relative biomass during spring and fall by 2050 across the Northeast continental shelf under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020), and larger increases in relative biomass are projected in the Gulf of Maine, while losses are projected in the Southern New England – Mid-Atlantic region. Six Horseshoe Crab populations in the Gulf of Maine, Mid-Atlantic, southeastern Atlantic US, Florida Atlantic, Northeast Gulf of Mexico, and Yucatan Peninsula of Mexico are genetically distinct (King et al., 2005 & 2015; Able et al., 2019; Smith et al., 2009, 2017, & 2023), with Smith et al. (2023) partitioning the species into 14 spatial conservation units. Within the Gulf of Maine, three genetically distinct subpopulations are embayment-
centered north and south of Great Bay, New Hampshire, and in the vicinity of Hog and Taunton Bays in Maine. Horseshoe Crabs are year-round residents in the estuaries and bays of the Northern Gulf of Maine, with little demographic exchange between the three spatial units that limits the potential for adult migration to recolonize an extirpated population. Recolonization is also unlikely to occur by larval dispersal because larvae tend to stay near their natal beaches (Botton and Loveland, 2003; Smith et al., 2023).

Populations within the Mid-Atlantic, from Cape Cod, Massachusetts, to Shackleford Banks, North Carolina, are genetically distinct from Gulf of Maine populations. Smith et al. (2023) divided Mid-Atlantic Horseshoe Crab into three spatial conservation units based on management programs and distinct harvest programs, with the Northeast spatial unit extending from Cape Cod to Connecticut, the New York spatial unit including Long Island Sound and the New York Bight, and the Delaware Bay spatial unit from New Jersey south of Raritan Bay to North Carolina. Hallerman et al. (2022) found genetic and demographic mixing within the Mid-Atlantic population from Massachusetts to South Carolina, including Delaware Bay. An increase in average water temperature over the next century may negatively impact the southernmost extent of the range of the species, leading to range shifts or jeopardizing their viability (Smith et al., 2023).

Horseshoe Crabs exhibit variable seasonal inshore-offshore migration patterns depending on the subpopulation and, further, migration may vary with age and sex of Horseshoe Crab as well, with estuaries and their intertidal beaches and tidal flats serving as both spawning habitat and juvenile nurseries (Smith et al., 2009; Able et al., 2019; Botton et al., 2021; Colon et al., 2022). Cheng et al. (2022) found that water temperature is likely one of the primary environmental cues for crab spawning migration, while salinity appeared unimportant. Smith et al. (2023) found that Horseshoe Crabs may adapt to rising sea levels by spawning in deeper waters, which could significantly change phenology and trophic relationships. The availability of spawning and juvenile nursery habitats (e.g., sandy tidal flats) with sea level rise is dependent on the extent of coastal development and shoreline armoring (Loveland and Botton, 2015; Able et al., 2019; Smith et al., 2023).

The viable range expansion of Horseshoe Crab is dependent on the availability of suitable habitat, with the greatest habitat losses anticipated to be due to coastal squeeze where beach habitat along the shoreline cannot migrate landward due to coastal development (Loveland and
Botton, 2015; Wisnewski and Tanacredi, 2022; Smith et al., 2017 & 2023). Although Horseshoe Crabs can spawn in mud or cobble habitats where the preferred sandy beaches are unavailable, these habitats are suboptimal (Smith et al., 2023). The egg development is highest where intertidal sediments are moist and well-oxygenated, most often in the highest high intertidal zone, which is accessible during spring or full moon high tides (Vasquez et al., 2015a & 2015b; Cheng et al., 2016; Smith et al., 2009 & 2023). Sea level rise may also affect important submerged aquatic vegetation larval and juvenile habitat (Botton et al., 2022; Colon et al., 2022; Smith et al., 2023).

Shifts in Phenology

The spawning phenology of Horseshoe Crabs is expected to shift with climate change because spawning cues are related to temperature and photoperiod (Cheng et al., 2016 & 2022; Smith et al., 2023). Horseshoe Crabs tend to migrate up into estuaries in the spring in search of warmer waters for spawning, and their preferred spawning sites are in the northern portion of the range in the Gulf of Maine (Cheng et al., 2016 & 2022). At the southern end of the range, spawning is triggered by cooling waters, so temperature increases could reduce reproductive activity and create suboptimal conditions for egg development and juveniles at the southern end of the range (Smith et al., 2023). To adapt to rising sea levels, Horseshoe Crabs may start spawning in deeper waters, which could result in significant changes to phenology and trophic relationships (Smith et al., 2023).

Horseshoe Crab migration phenology may be disrupted by more frequent storms due to climate change, reducing egg survival due to increased freshwater flooding nursery and juvenile habitat (Smith et al., 2017 & 2023). Phenological shifts in the timing or distribution of spawning in some areas could lead to mismatches with interspecies relationships, such as migrating shorebirds (Smith et al., 2017 & 2023).

Morphological or Physiological Shifts

Spawning and growth of Horseshoe Crabs varied regionally as a function of temperature (Vasquez et al., 2015a & 2015b; Cheng et al., 2016 & 2022; Botton et al., 2022; Smith et al., 2023). Vasquez et al. (2015b) found that high temperature, low oxygen, or low salinity individually can reduce developmental success by up to 72% in Horseshoe Crab eggs, but
combinations of these individual stressors led to complex interactions and stronger effects. For example, low oxygen combined with high temperature was lethal to Horseshoe Crab eggs, and increases in ocean temperatures and precipitation events that affect local salinity due to climate change may amplify these stressors and reduce fitness (Vasquez et al., 2015b). If Horseshoe Crabs experience developmental delays and reduced fitness due to these environmental stressors, their larva may not emerge from the sand during biweekly spring tide events and could be more vulnerable to predation by remaining in the sand for longer periods (Vasquez et al., 2015b).

**Population Changes**

Horseshoe Crab populations are likely most threatened by overharvesting and habitat loss, the latter amplified by climate change (Botton et al., 2022; Smith et al., 2023). Botton et al. (2021 & 2022) found the greatest threat to the species is the loss of high-quality spawning habitat. Population growth can be limited by the availability of optimal spawning habitat, which is undisturbed and well-oxygenated sandy beaches (Smith et al., 2017; Botton et al., 2021). Vasquez et al. (2015a) determined that spawning Horseshoe Crabs selected optimal nest sites on high tidal beaches to minimize exposure of eggs to environmental stressors. When exposed to temperatures, oxygen levels, and hydrogen sulfide levels typical of low tidal beaches, developmental success was reduced by up to 25% (Vasquez et al., 2015a). Coastal development and associated hard shoreline engineering structures can truncate the sandy beach below the spring high tide line, constricting the amount of sandy beach available for spawning and potentially leading to the mortality of spawning Horseshoe Crabs that become trapped (Smith et al., 2017). Further, beach erosion can expose peat or mud substrates, which typically have anoxic and low-oxygen conditions unsuitable for egg development (Smith et al., 2017). The potential long-term recovery of the species will depend on the mitigation of cumulative habitat losses from coastal development and sea level rise (Botton et al., 2022; Smith et al., 2023). On Long Island, New York, the annual loss of breeding habitat is approximately 3% (Wisnewski and Tanacredi, 2022).

The rate of Horseshoe Crab population growth is highly sensitive to egg development and survival of their early life stage, which suggests that a shift to suboptimal spawning and nursery habitats as a result of sea level rise would reduce the carrying capacity of habitat and further limit potential recovery of the species (Smith et al., 2023). Smith et al. (2023) concluded that the
Using the Green Status of Species assessment process from the International Union for the Conservation of Nature (IUCN), Smith et al. (2023) developed climate model projections for future population viability and ecological functionality for 14 Horseshoe Crab population units and found the Northern Gulf of Maine population is likely to remain viable for the next 10 years without harvesting but the small populations that do not mix create vulnerability to catastrophic events. Over the next century, the Northern Gulf of Maine population of Horseshoe Crabs is expected to remain ecologically functional but face significant habitat loss due to sea level rise and coastal squeeze (Smith et al., 2023). Further, they found that the Cape Cod to Connecticut subpopulation was currently stable with conservation measures mitigating harvest pressure and habitat loss, but reducing harvest pressure would enhance population stability. In the long term, populations are likely to be viable, but their ecological functionality remains uncertain due to uncertainty in climate change impacts (Smith et al., 2023). The New York subpopulation was found to be similar, though additional harvest restrictions may be needed in the short term and may experience uncertain long-term viability due to climate change (Smith et al., 2023). There are concerns that the Delaware Bay population faces further population declines and even collapse due to the interactive effects of harvest and climate without restrictive fishery management actions (Smith et al., 2023).

Overall, the long-term recovery of Horseshoe Crab is expected to have widespread habitat loss that will impact ecological functionality range-wide. The long-term viability of Horseshoe Crab on the Atlantic and Gulf Coasts has significant uncertainties, with extirpations possible in the Florida Atlantic and Yucatan Peninsula regions due to the combined habitat loss effects of climate change and anticipated coastal development (Smith et al., 2023). The Northern Gulf of Maine subpopulation is one of only two of the 14 subpopulations expected to retain ecological functionality over the next century, with the other in the Southeast (South Carolina and Georgia). Smith et al. (2023) conclude that current conservation concepts for the species are not expected to mitigate habitat loss at the scale needed to restore ecological functionality range-wide. They found that conserving critical habitats for spawning and early life stages is needed to achieve long-term ecological functionality in the face of climate change.
Indirect Effects

Horseshoe Crabs are threatened by sea level rise and severe weather events due to climate change, with some regional populations vulnerable to extirpation due to declines in small and vulnerable populations (Botton et al., 2021 & 2022; Smith et al., 2017 & 2023). New habitat may be created locally by sea level rise and an increase in storm frequency and intensity, but in other areas beaches are expected to erode or be lost to coastal squeeze. The northern Gulf of Maine population of Horseshoe Crabs is threatened by significant spawning habitat losses because of the limited availability of sandy beach habitat. Sandy beaches in the northern Gulf of Maine tend to be narrow and backed by rocky shorelines and, therefore, are more likely to be flooded with sea level rise (Smith et al., 2023). The Mid-Atlantic populations of Horseshoe Crab are currently more threatened by development, pollution, and harvest as bait and the medical industry than by climate change, but the loss of high-quality beach habitat for egg development due to sea level rise and coastal squeeze will likely lower carrying capacity in the long-term (Smith et al., 2023).

Bay Scallop
Changes to Morphology or Physiology

Bay Scallops are particularly sensitive to ocean acidification during the larval life stage, with legacy consequences for their later life stages. Exposure to ocean acidification has the highest impact during the earliest larval stage, with negative impacts on growth and survival observed in laboratory experiments in as little as one day or less (Gobler and Talmage, 2013; White et al., 2013 & 2014). Exposure to elevated CO₂ levels during fertilization decreased survival rates of larval Bay Scallops in an experiment conducted by White et al. (2014), with results indicating that the first two hours are a critical exposure window for ocean acidification. Gobler and Talmage (2013) found that high CO₂ concentrations resulted in cascading negative impacts on Bay Scallop physiology, partly due to lower calcification rates for their shells. Survival rates declined significantly with exposure to elevated CO₂ levels during the first four days of larval development. However, a ten-day exposure later in the larval development period did not reduce survival, and individuals who survived exposure to high CO₂ levels during larval development had higher juvenile growth rates when exposed to normal CO₂ levels when compared to individuals who were not exposed as larvae to high CO₂ levels (Gobler and Talmage 2013). Scallops that were exposed to even moderate CO₂ levels during larval
development were smaller than those grown with normal CO₂ levels (Gobler and Talmage, 2013; White et al., 2013 & 2014), with legacy effects observed at ten months of age (Gobler and Talmage, 2013). Increased juvenile growth rates did not overcome these size differences, suggesting potential population-level effects from ocean acidification (Gobler and Talmage, 2013). White et al. (2014) also found shell deformities in scallops exposed in the laboratory to continuous high CO₂ levels when compared to those only exposed after two hours post-fertilization.

Experiments by Young and Gobler (2018) observed reduced growth rates in shell and/or tissue growth of Bay Scallops and three other estuarine bivalves with exposure to ocean acidification but found that the presence of the macroalgae sea lettuce (*Ulva* spp.) reduced the degree of impact from the ocean acidification. They also documented higher calcium carbonate saturation states when *Ulva* spp. was present with and without ocean acidification conditions, suggesting that photosynthesis and/or nitrate assimilation by the macroalgae increased alkalinity levels. Young and Gobler (2018) suggested that large collections of macroalgae, either naturally occurring or aquaculture collections, could provide refuges for bivalves to mitigate the negative effects of ocean acidification.

Xing et al. (2016) tested the thermal tolerance of Bay Scallops at different life stages, finding evidence that indicated that younger and smaller scallops have higher thermal tolerances than older and larger scallops and that spawning behavior by adults may decrease thermal tolerance. Some populations of Bay Scallops in China have been observed to have higher tolerance to thermal stress than others, spurring research efforts into underlying genetic markers (Yang et al., 2013; Yang et al., 2014; Yang et al., 2015; Zhu et al., 2021; Liu et al., 2022). The *A. i. irradians* subspecies exhibited more sensitivity to thermal stress than the *A. i. concentricus* subspecies, the latter of which appeared to have more thermal adaptation (Yang et al., 2014; Yang et al., 2015; Zhu et al., 2021; Liu et al., 2022).

In estuaries, Bay Scallops are more sensitive to climate change stressors such as intermittent hypoxia compared to Eastern Oysters (*Crassostrea virginica*), Hard Clams (*Mercenaria mercenaria*), and Blue Mussels (*Mytilis edulis*), although those bivalves also exhibit negative physiological effects to varying degrees (Clark and Gobler, 2016; Ivanina et al., 2016; Ivanina and Sokolova, 2016; Young and Gobler, 2018). Bay Scallops also showed evidence of oxidative stress and cellular damage but did not lead to a severe deficiency in energy
levels.

Eutrophic estuaries commonly experience seasonal patterns of diurnal fluctuations in dissolved oxygen and pH levels (Clark and Gobler, 2016; Gobler et al., 2017; Yang et al., 2021a). Yang et al. (2021a) found that dissolved oxygen is significantly depleted during summer in both surface and bottom waters due to temperature, microbial respiration, and photosynthesis drivers. Seasonal temperature was the most important driver for depleting dissolved oxygen. Photosynthesis could offset some but not all temperature-induced deoxygenation in surface waters. In bottom waters, biological respiration plays more of a role with temperature drivers. Yang et al. (2021a) concluded that seasonal hypoxia and acidification in the coastal waters of the Bay Scallop habitat are highly associated with the stratification of the water column due to temperature with complex environmental processes.

When Bay Scallops are exposed to continuous acidification or hypoxia conditions, larval survival, growth, and development declined. Combined exposure to acidification and hypoxia resulted in negative additive effects (Clark and Gobler 2016). However, juveniles were more tolerant to acidification exposure than larvae (Gobler et al., 2017). If exposure was diurnal, the return to normal conditions was not long enough to reverse the physiological stress, and/or the severity level of the hypoxia and acidification was too intense to overcome fully (Clark and Gobler 2016). When acidification and hypoxia in combination had diurnal variation, larval Bay Scallop survival was significantly lower than when levels were held constant (Gobler et al., 2017). These results have led to the conclusion that these interactive effects lead to higher vulnerability and can be a significant threat to Bay Scallop larvae with potential population-level impacts as these conditions intensify in the future (Clark and Gobler, 2016; Gobler et al., 2017).

Tomasetti et al. (2023) assessed the vulnerability of Northern Bay Scallops (*A. irradians*) in the Northeast to hypoxic and thermal stress in combination. Most of the geographic range of Northern Bay Scallop experienced significant warming during the summer from 2003 to 2020. Tomasetti et al. (2023) documented the mortality of a local population of Northern Bay Scallops in New York during an eight-day heatwave in their estuary that coincided with a severe diel-cycling hypoxia period. At the same time, a population in Massachusetts with similar dissolved oxygen levels but lower mean daily temperatures did not suffer a die-off. Subsequent laboratory experiments sought to recreate these observed conditions and found an increase in the likelihood of mortality by 120 times with the combined effects of high temperatures and hypoxia.
Field experiments in New York and Massachusetts found that higher daily temperatures and low dissolved oxygen levels negatively impacted Bay Scallops' oxygen consumption rates and aerobic activity. Overall, these results provide strong evidence that the combined effects of hypoxic and thermal stress on Bay Scallops negatively impact their physiology and survival with potential population-level effects.

Gobler et al. (2014) tested the combined effects of low oxygen and ocean acidification on the early life stages of Bay Scallop and Hard Clam. Scallop larval survival was reduced by more than 50% with ocean acidification alone, and low oxygen reduced growth and metamorphosis by more than 50% alone. The combined effects of both ocean acidification and low oxygen were additive and negative, leading Gobler et al. (2014) to recommend that climate change stressors to Bay Scallops (and Hard Clams) should be considered in combination and not individually to predict species responses.

Stevens and Gobler (2017) subsequently tested the individual and interactive effects of ocean acidification, thermal stress, and hypoxia on four marine bivalves – Bay Scallop, Oyster, Hard Clam, and Blue Mussel (*Mytilus edulis*). Higher temperatures resulted in the most consistent physiological responses of the bivalves, but the responses were both negative and positive. Bay Scallops were the most sensitive of the four species, with reduced survival, shell growth, and/or tissue weight when exposed to either low dissolved oxygen or elevated acidity. Hypoxia exposure resulted in significantly increased rates of respiration in Bay Scallops. The interactive effects of low oxygen and acidification often were antagonistic and resulted in higher growth rates than they would individually, suggesting a potential anaerobic metabolic response. The interactive effects of acidification and higher temperatures were antagonistic and synergistic (Stevens and Gobler, 2017).

Population Changes

Griffith and Gobler (2017) found that the negative impacts of ocean acidification on Bay Scallops are transferred to their offspring, with offspring significantly more vulnerable to not only ocean acidification effects but to other stressors like thermal stress, exposure to harmful algae, and limitations in food resources as well. They conclude that Bay Scallops are not likely to acclimate to ocean acidification in the short term and that with continued ocean acidification that
the negative impacts to scallop populations will compound and become more severe over time (Griffith and Gobler 2017).

Grear et al. (2020) developed population models for Bay Scallops and Hard Clams to evaluate the population-level impacts of acidification on the bivalves. The cumulative risk of extinction was higher for Bay Scallops than for Hard Clams. The five-year risk of Bay Scallop extinction ranged from 56% to greater than 99% with increasing acidification levels (Grear et al., 2020).

**Indirect Effects**

Griffith et al. (2019) documented how the presence of two algae known to cause harmful algal blooms, *Cochlodinium polykrikoides* and *Aureococcus anophagefferens*, interacted with ocean acidification and higher temperatures negatively impacted the larval physiology of Bay Scallops in their synthesis of protein, shell growth, cellular metabolism, and membrane transport. With ten days of exposure, each harmful algae decreased larval survival at levels significantly higher than the individual stressors (Griffith et al., 2019).

Population declines due to the loss of juvenile estuarine habitat for Bay Scallops are amplified by the impacts of ocean acidification, which decrease larval survival and shell size (White et al., 2014). The impacts of ocean acidification may be amplified by intensive shellfish aquaculture (Yang et al., 2021b). Yang et al. (2021b) found that scallop calcification and mixing were the main drivers of seasonal variation in alkalinity levels. The key environmental drivers for levels of dissolved inorganic carbon were mixing, exchange between the air and water, and microbial activity, such as microbial respiration processes and photosynthesis. The combined effects of temperature, air-sea exchange, scallop metabolic activities, microbial activity-controlled pH, pCO$_2$, and aragonite saturation state levels. They found that large-scale scallop farming significantly increases the ratio of dissolved inorganic carbon to alkalinity, reducing the total alkalinity concentration and, thus, the buffering capacity of the carbonate system. The growth of Bay Scallop shells in intensive aquaculture removes carbonate from the seawater, likely accelerating ocean acidification at the local level (Yang et al., 2021b).
Atlantic Sea Scallop

Shifts in Range or Depth

Atlantic Sea Scallops are a sentinel species for climate change in marine systems (Stokesbury and Bethoney, 2020). They range from Cape Hatteras, North Carolina, north to the Gulf of St. Lawrence, residing on the continental shelf in nearshore waters that typically range from 0℃ to 17℃ in temperature, with optimal growth between 10℃ and 15℃ and temperatures above 21℃ lethal. Atlantic Sea Scallops are typically found between 15 and 110 meters in depth but can be found in waters as shallow as 2 meters in the northern part of its range and are uncommon deeper than 60 meters (Cooley et al., 2015; Torre et al., 2018; Stokesbury and Bethoney, 2020). Their preferred salinity is full-strength seawater of 35%, with salinities lower than 16.5% lethal (Torres et al., 2018). Individuals reach ages of at least 18-20 years, with populations in the Mid-Atlantic exhibiting slightly faster growth rates than those in the Gulf of Maine (Cooley et al., 2015) and populations in the northern portion of the range showing smaller average shell sizes (Stokesbury and Bethoney, 2020). Warming bottom ocean temperatures with climate change may exceed Atlantic Sea Scallops' thermal tolerance in the current range's southern portion or shallower habitats (Cooley et al., 2015). Adult Atlantic Sea Scallops are generally sedentary, with their distribution dependent on larval recruitment (Cooley et al., 2015; Torre et al., 2018; Stokesbury and Bethoney, 2020).

The geologic record suggests that Atlantic Sea Scallops shifted their range on the continental shelf multiple times as sea levels rose and fell and water temperatures changed (Stokesbury and Bethoney, 2020). Between 1974-1977 and 2019-2022, Atlantic Sea Scallops' spring range shifted 1.52 degrees (168.83 km) north and contracted by 0.4 degrees (44.95 km). During the fall season, Atlantic Sea Scallops shifted 0.83 degrees (92.5 km) north and contracted their range by 0.31 degrees (34.69 km) between the periods of 1974-1976 to 2019-2022. Atlantic Sea Scallops shifted 12.8 meters and 9.9 meters deeper during spring and fall, respectively (NOAA Fisheries, 2022).

Model projections of future range shifts of Atlantic Sea Scallops in the Northeast estimate the population will move northward over the next 80 years in association with anticipated changes to bottom temperature and salinity while experiencing a significant loss of relative biomass during fall by 2050 across the Northeast continental shelf under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). However, small increases in relative
biomass are projected in the Gulf of Maine during spring and fall, while losses are projected during fall in the Southern New England – Mid-Atlantic region.

At the northern portion of the range, Atlantic Sea Scallop populations in the Bay of Fundy are positively correlated with temperature, which is thought to lead to rapid development of larvae and improved survival of juveniles and adults (Dickie, 1955; Caddy, 1979; Torre et al., 2018). Torre et al. (2018) developed a bioclimate envelope model for Atlantic Sea Scallops to evaluate habitat suitability with climate change in the Gulf of Maine from Massachusetts to Maine. The authors found peak habitat suitability indices were a high annual bottom temperature of 15°C for inshore habitats and 10°C for offshore habitats, a lowest annual bottom salinity of 31% inshore and 33% offshore, a depth of 10 meters inshore and 37 meters offshore, and an average current speed of 0.05 meters per second inshore and 0.1 offshore. The authors also found for inshore areas, bottom temperature, bottom salinity, and flow velocity were the most important environmental variables, while bottom temperature, bottom salinity, and depth were the most important in offshore areas. Changes in habitat suitability with changing climatic conditions from 1978-2013 showed increasing habitat suitability in inshore areas and decreasing habitat suitability in offshore areas, with the exception of offshore shoal areas, which exhibited an increasing trend similar to inshore areas of the Gulf of Maine. Torre et al. (2018) suggested that changing climatic conditions have increased the habitat suitability of inshore areas in the Gulf of Maine while offshore habitat suitability remained relatively stable from 1978 to 2013. Asci et al. (2018) also found that offshore populations of Atlantic Sea Scallops in the central Gulf of Maine did not change significantly between 1986 and 2014, suggesting that the benthic communities of offshore shoals experience high natural disturbance rates that could increase their resiliency to fishing activities and environmental drivers.

Changes to Morphology or Physiology

Over the past 150 years, ocean acidification has increased by 26%, with additional acidification of two to three times that rate projected for the next century (Cooley et al., 2015). Increased ocean acidity affects the sea surface saturation state (or solubility) of aragonite, the form of calcium carbonate commonly found in larval bivalve shells. The calcium carbonate saturation state of the continental shelf decreases from south to north, offshore North Carolina to New Hampshire, in both summer and winter, with winter saturation states lower than summer.
Ocean acidification increases the solubility of calcium carbonate (or decreases the saturation state), which can reduce the growth and energy budgets of marine and estuarine bivalves (Cooley et al., 2015; Cameron et al., 2022). Warming ocean temperatures, on the other hand, can increase growth by increasing metabolism but only up to a limiting high-temperature tolerance, above which growth can be impeded (Cooley et al., 2015).

Cameron et al. (2022) measured the effects of ocean acidification and water temperature on the calcification rate, carbonate chemistry of the extrapallial fluid, respiration, and survival of Atlantic Sea Scallops. Elevated ocean acidity inhibited calcification and respiration. The synergistic effects of higher ocean acidity with high water temperature resulted in mortality. They suggest that growth and survival declines were likely a result of dissolution of the external shell, thermal stress, and a reduced metabolism from ocean acidification. The negative physiological effects of ocean acidification on the Atlantic Sea Scallops' condition were not mitigated by current harvest regulations (Cameron et al., 2022).

Liu et al. (2021) modeled the inter-annual variation of scallop conditions on Georges Bank with sea surface temperature and chlorophyll-a concentration from 1985 to 2019. Their results found a positive correlation between scallop conditions in May with sea surface temperature variability in the preceding winter-spring months. The relationship between chlorophyll-a concentration and scallop condition was weak, likely due to vertical mixing of the water column that transports phytoplankton to its benthic habitat (Liu et al., 2021).

The energetic cost of calcification of shells is thought to increase with ocean acidification, with reduced energy budgets for reproduction or immunity (Cooley et al., 2015). A model for the level of energy available for growth for Atlantic Sea Scallops found that energy available for growth on the continental shelf of the Northeast US is spatially heterogeneous, with higher levels in May-June and lower levels in January-February (Zang et al., 2022). The authors also found that thermal stress in the Mid-Atlantic Bight resulted in negative energetics for growth from July to October and food availability from particulate organic matter was higher in the cold seasons and lower in the warm seasons, which, when combined with the thermal stress of the warm season for the Mid-Atlantic Bight, led to a reduction in suitable habitat. No synergistic effects of warming and food deficiency were found on Georges Bank (Zang et al., 2022). The model also found sensitivities for increasing temperature and food availability with
scallop size, indicating that larger scallops are more sensitive and that suitable habitats may decrease as scallops grow older or bigger (Zang et al., 2022).

Changes in Population

Stokesbury and Bethoney (2020) conducted a range-wide survey to quantify the abundance and distribution of Atlantic Sea Scallops from 2016 to 2018, estimating a total population of 34 billion individuals from North Carolina to Newfoundland. They found Georges Bank had the highest population density, providing 71% of the overall species abundance, while the Mid-Atlantic contained 27% of the total population, and the Gulf of Maine and the northern range hosted the remaining 2% of the population. The naturally high climate variability in the range of the Atlantic Sea Scallop is anticipated to increase with future climate change. Large areas of the Georges Bank and Mid-Atlantic exhibited high productivity and were self-sustaining, whereas the northern grounds and the Gulf of Maine may be reproductively isolated (Stokesbury and Bethoney, 2020). Evidence suggests that recruitment patterns and extreme recruitment events may generate larval recruitment from highly productive Georges Bank and the Mid-Atlantic aggregations to the more isolated Northern grounds and Gulf of Maine (Stokesbury and Bethoney, 2020). If larval recruitment and settlement of Atlantic Sea Scallops can be widespread during certain years, this suggests some adaptive capacity to respond to changing climatic conditions.

Ocean acidification could affect larval recruitment of Atlantic Sea Scallops (Cooley et al., 2015), but populations do not appear to be limited by larval supply in Georges Bank currently (Cooley et al., 2015; Stokesbury and Bethoney, 2020). Recruitment in Georges Bank could be limited by suitable settlement habitat, predation, or food availability. In the Mid-Atlantic, however, larval recruitment may be limited by larval supply, and a reduced larval supply due to ocean acidification in the future could reduce recruitment significantly in that region (Cooley et al., 2015). Periodic extreme recruitment events may not reach their full potential if warmer waters amplify the impact of fishing bycatch, such as what occurred in 2003 in the Mid-Atlantic when 10.4 billion juvenile Atlantic Sea Scallops (more than half of the year class) died as bycatch because they were raised to the surface through water temperatures above the lethal limit and then exposed to high air temperatures for several hours before being returned to the sea floor (Stokesbury et al., 2011).
Over the next several decades, increasing bottom ocean temperatures and ocean acidification due to climate change are anticipated to threaten Atlantic Sea Scallop populations increasingly. Warming ocean temperatures could increase their growth rates, while ocean acidification could slow growth and decrease recruitment (Cooley et al., 2015). Cooley et al. (2015) developed an integrated assessment model to simulate Atlantic Sea Scallop populations in the near- and long-term under different carbon dioxide emissions scenarios. Their model results indicate that Atlantic Sea Scallop populations could significantly decline by 2050 under the current emissions scenario (RCP 8.5), assuming current harvest regulations, decreased recruitment and slower growth due to ocean acidification, and increased growth due to warming oceans. An increased growth rate due to ocean warming appears to outweigh the negative impacts of ocean acidification until 2030 when the pattern reverses, and the net impacts are negative growth rates for Atlantic Sea Scallops (Cooley et al., 2015). With further increases in temperature and ocean acidification, the impacts of commercial harvest will be amplified, and current harvest regulations will become out of sync with the population size distribution (Cooley et al., 2015).

Model projections of the impacts of future climate change and harvest management of Atlantic Sea Scallop with a focus on ocean acidification estimated biomass reductions of 13% by the end of this century with lower emissions scenarios coupled with high harvest impact scenarios (Cooley et al., 2015; Rheuban et al., 2018). Under high emissions scenarios, Atlantic Sea Scallop biomass may be reduced by more than 50% by the end of this century. Commercial harvest of the species amplifies the impact of ocean acidification and vice versa. Harvest limits can mitigate some of the anticipated ocean acidification impacts, with a 10% increase in harvest closure areas increasing biomass by more than 25% under the highest impacts of the ocean acidification scenario. A long-term population decline due to ocean acidification is projected regardless of the harvest management scenario (Rheuban et al., 2018).

Summary

Marine bivalves are particularly vulnerable to ocean acidification due to their carbonate shells, with much of the literature focused on this aspect of climate change either alone or in combination with other stressors like elevated temperatures or dissolved oxygen. These stressors, both individually and in synergy, have documented physiological, morphological, and survival
effects on both Bay Scallops and Atlantic Sea Scallops. The larval life stage appears to be the most vulnerable life stage for exposure to climate change stressors (Gobler and Talmage, 2013; White et al., 2013; White et al., 2014; Clark and Gobler, 2016; Gobler et al., 2017).

Much of the climate change research for Atlantic Sea and Bay Scallops has focused on the effects of ocean acidification and increasing water temperatures, with little known about the impacts of these other large-scale marine habitat changes on the species. Some of the research was from the fisheries perspective with focused questions relevant to stock management, such as the selective breeding of Bay Scallops for thermal tolerance (e.g., Yang et al., 2015; Zhu et al., 2021). Needed research to fill data gaps for these marine bivalves include population-level effects of ocean acidification (Rheuban et al., 2018), as well as the interactive effects of ocean acidification and warmer waters on physiology with benthic habitat disturbance and losses both in estuaries for Bay Scallops and offshore for Atlantic Sea Scallops. In addition, a better understanding of the effects of shifting or changing currents on spawning, recruitment, and settlement and the effects of increased frequency and intensity of coastal storms and extreme weather events on Bay Scallop populations and habitat is needed.

B) BIRDS
Table 2.4. RSGCN bird species and number of scientific articles cited in this report for each species.

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<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
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**Figure 2.3.** Number of scientific articles for all RSGCN birds by biological response type.

**Overview**

Coastal shorebirds, waterbirds, waterfowl, and seabirds face multiple climate change impacts. There is increasing evidence for phenological mismatches and changes to habitat availability, quality, and distribution. Recent studies suggest that the severity of impacts varies with life history traits, phenological plasticity, and type of migration. Arctic-breeding, long-distance migrant birds such as Pale-bellied Brant, Rufa Red Knot, and Purple Sandpiper rely on syncing migration with environmental conditions and food resources.
The degree of existing or potential phenological mismatch due to climate change depends on the type of migration. For example, long-distance, non-stop migrants such as Pale-bellied Brant, rely on cues at their spring staging areas; these areas may become out of sync with spring onset in the northern breeding areas where warming temperatures are higher (latitude-dependent) and vary regionally (Clausen and Clausen, 2013). Short-distance migrants are more likely to remain in sync with seasonal environments because they move across shorter distances with relatively similar conditions (Clausen and Clausen, 2013). Species that make migratory stopovers are more likely to acclimate to environmental cues and adjust migratory patterns accordingly (Clausen and Clausen, 2013).

Across all taxa, species more specialized in resource use will be more vulnerable to changing prey and habitat availability than generalists. While not unique to just migratory birds, there are widespread concerns for fitness consequences resulting from phenological mismatches in food resources, particularly during breeding and migration, for many coastal bird species (Catry et al., 2013; Nightingale et al., 2018; Taillie and Moorman, 2019; Tucker et al., 2019; Bratton et al., 2022). Moreover, strong site fidelity during the chick-rearing period and a limited radius to hunt for food increases the vulnerability of coastal birds to changes in local environmental conditions and human disturbances, including wind energy development, recreation, and fishing (Gibson et al., 2018; USFWS, 2020a; Bratton et al., 2022).

Coastal birds are threatened by “coastal squeeze” of habitat from sea level rise, whether that be on their Arctic tundra breeding grounds of Red Knot and Pale-bellied Brant (Clausen and Clausen, 2013; Clausen et al., 2013a; Smith et al., 2020), rocky intertidal foraging habitat of Purple Sandpiper (Nightingale et al., 2018), salt marsh and tidal wetland habitat of Eastern Black Rail or Black Skimmer (Adam, 2002; Pontee, 2013; Hunter et al., 2015; Roach and Barrett, 2015; Tallie and Moorman, 2019; Tattoni et al., 2020), or sandy beach nesting, foraging, or roosting habitat of Piping Plover, Red Knot, American Oystercatcher, Black Skimmer, Common Tern, or Least Tern (Seavey et al., 2011; Doody 2013; Sims et al., 2013; National Wildlife Federation and Manomet Center for Conservation Sciences, 2014; Burger and Niles, 2014; Ivajnišč et al., 2017; Maslo et al., 2018; Tattoni et al., 2020).

Coastal beach habitats that were previously able to migrate across the landscape in response to sea-level changes are now impeded by coastal modifications and face additional habitat loss and degradation from the direct and indirect effects of climate, including erosion and
invasive plants (Taillie and Moorman, 2019). Coastal marshes can adapt to rising sea levels through vertical accretion or marsh migration, where conditions allow, but adjacent forests where the saltwater resistance of mature trees limits the pace of migration can lead to net losses, particularly for high marsh habitats (Field et al., 2016; Taillie and Moorman, 2019). As sea levels rise at an accelerating rate, these coastal habitats often narrow in spatial extent and are at high risk of becoming completely inundated at some locations due to interacting and amplifying climate and non-climate stressors.

Increased storm frequency and intensity across the region can create or enhance the early successional beach habitat used by shorebirds and waterbirds for short periods of time (Schulte and Simons, 2015 & 2016; Robinson et al., 2019; Walker et al., 2019; Zeigler et al., 2019). In some areas, active management of coastal wetland vegetation and natural disturbance regimes can increase the availability of high-value food or habitat resources and partially offset reductions in resource availability or increase the carrying capacity of habitats (Clausen et al., 2013a; Livolsi et al., 2021; Maslo et al., 2018 & 2019; Robinson et al., 2019; Walker et al., 2019; USFWS, 2020a). However, the cumulative effects of coastal squeeze and increased storm impacts on habitat availability, suitability, and distribution across the geographical ranges of coastal birds are of growing concern. Spatially explicit projections of where the coastal squeeze is happening the fastest overlayed with sea level rise projections are useful to identify local areas to prioritize actions such as assisted migration and habitat restoration.

The community composition of coastal birds in the Northeast is generally shifting towards warmer-breeding coastal bird species, resulting in declines in breeding habitat availability and increased interspecific competition due to the influx of Arctic- and southern-nesting shorebirds into the area (Anderson et al., 2023).

Disease and pathogens are another emerging and indirect threat of concern that can result from climate-related increases in storm and precipitation frequency and intensity. For example, heavy rainfall and hurricane events in Tampa Bay, Florida, during the 2016 breeding season for Black Skimmers resulted in repeated sewage overflows into the estuaries adjacent to a breeding colony (Shender et al., 2022). The resulting Salmonellosis killed 39% of the Black Skimmer fledglings in the colony because harmful pathogens persisted in the beach sand, allowing the substrate to harbor reservoirs of bacterial pathogens (Whiley et al., 2018; Shender et al., 2022). While this example is outside of the Northeast, it identifies potentially hazardous conditions to
monitor for in Northeastern habitats as the region continues to experience wetter and warmer conditions.

**American Black Duck**

*Shifts in Range, and Elevation, or Depth*

Across multiple climate change scenarios, American Black Duck populations are projected to increase by the end of the twenty-first century (2071-2100; Adde et al., 2020). Projections indicate a northward and eastward shift of the core abundance of breeding American Black Ducks in eastern Canada with no time lag since the ducks are ground nesters, as compared to cavity-nesting waterfowl that face a multi-decade time lag for their needed forested habitat characteristics to shift (Adde et al., 2020).

Over the past half-century, the wintering range of American Black Ducks shifted northward in association with warming temperatures (LaSorte and Thompson, 2007; Meehan et al., 2021). While the overall annual population change in wintering American Black Ducks had no detectable trend at the continental scale, there was spatial variation in declines, with positive abundance trends in colder regions and negative trends in warmer regions (Meehan et al., 2021). Future projections estimated that with a 3°C increase in global annual temperature, the wintering range of American Black Ducks will continue to shift north, decreasing across the warmer Mid-Atlantic, Gulf, and Mississippi Alluvial Valley states (Bateman et al., 2020).

*Shifts in Phenology*

An analysis of the migration phenology of American Black Ducks found high fidelity along the Atlantic and lower fidelity to the Mississippi Flyway, particularly for Canadian ducks (Lavretsky et al., 2014). Evidence also suggests two stocks of American Black Ducks may differ in landscape use, with eastern ducks on the Atlantic Flyway generally in their wintering habitat in December (Lavretsky et al., 2014). Peak fall migration between 1922 to 2007 found Western (Mississippi Flyway) ducks have longer and more variable migration patterns between November through February, with western ducks in the U.S. moving primarily from the Great Lakes to the Mississippi Alluvial Valley; approximately 20% of ducks moved later between the central and southern Atlantic coastlines and appeared to use a single route east (Lavretsky et al., 2014). Multiple migration routes were identified for American Black Duck in the eastern US,
with important wintering areas on Chesapeake and Delaware Bays and secondary concentrations along the coasts of New Hampshire and Massachusetts (Lavretsky et al., 2014).

Recent research identified a delay in fall migration of American Black Ducks of 3-5 days/decade or a shift of $18 \pm 6$ days for peak migration between 1968 and 2011 (Thurber et al., 2020). Peak migration date and northward range shifts were related to warmer winter temperatures and decreased snow cover (Thurber et al., 2020). Future projections estimate additional delays in the lower Great Lakes region of 12 to 19 days by 2050, which could result in American Black Ducks and Mallards (*Anas platyrhynchos*) overlapping on their overwintering grounds (Notaro et al., 2016) with concerns for decreased carrying capacity and increased competition for high-quality food resources (Schummer et al., 2020).

*Changes to Morphology or Physiology*

As sea level rise impacts gradually replace higher energy-density habitat types (e.g., high and low marsh) with lower-density ones (e.g., mudflat and subtidal communities), coastal impoundments managed for high-quality food resources can target increasing the proportion of habitat that maximizes the landscape's energetic carrying capacity (Livolsi et al., 2021). American Black Ducks in the Northeast have shown losses of body fat throughout the winter (Barboza and Jorde 2018). Birds in Maine had lower fat stores than birds in New Jersey, reflecting the higher energy demands of colder temperatures (Barboza and Jorde 2018). Variability in food availability increased the cost of both female and male ducks to maintain body mass (Barboza and Jorde 2018). The frequency of winter feeding or diet quality during spring did not affect egg production during the subsequent breeding season (Barboza and Jorde 2018). Populations in Maine had a narrower diet focused on marine invertebrates compared to populations in New Jersey, suggesting a diet shift at the northern edge of their wintering range (Barboza and Jorde 2018).

*Population Changes*

Despite evidence that most waterfowl exhibit an approximately balanced sex ratio at hatch, many North American ducks have a skewed adult sex ratio with a male bias (Ellis et al., 2022). In the Northeast, juvenile sex ratios for American Black Ducks from 1961-2015 appeared stable, but drought was correlated with decreased proportions of female juveniles (Ellis et al., 2022).
Indirect effects

Range shifts and increased population overlap can potentially increase competition for food resources in high-quality habitats. This has been observed on shared wintering grounds in New York, where Mallards exhibited aggression and behavioral dominance over American Black Ducks (Schummer et al., 2021).

American Oystercatcher

Population Changes

Projections indicate that Black Skimmers will lose critical nesting habitat due to the dual pressures of sea level rise and urban development (National Wildlife Federation and Manomet Center for Conservation Sciences, 2014; Seavey et al., 2011; Thorne et al., 2012; Sims et al., 2013, Ivajnšič et al., 2017, Maslo et al., 2018); in addition, increasing frequency and intensity of coastal storms and surges can create new habitat (Cohen et al., 2009, Schulte and Simons 2016; Woodrey et al., 2012) or have negative effects on nest survival (Schulte and Simons 2016). Predators can also compound the negative effects after major storms and impede recovery without management intervention. American Oystercatchers in North Carolina moved nests from the dune line onto new and expanded open sand flats following a hurricane (Burger and Niles 2014; Schulte and Simons 2015, 2016). Mammalian predators were a significant source of nest failure before the hurricane; however, nest survival and productivity increased when nests were located on open sand flats where they were less visible. It is also possible that the population of mammalian predators declined due to storm mortality (Schulte and Simons 2015, 2016).

Black Rail

Population effects

Models evaluating Black Rail occupancy across marsh to forest gradients suggest increases with greater Cladium jamaicense dominance and closer proximity to the forest-marsh interface where nests are less vulnerable to high water and flooding; however, results were highly uncertain due to low detection probability (Taillie and Moorman 2019). The occupancy of Black Rail and four other marsh bird species was greater in areas where conditions were
consistent with the effects of prescribed fire, such as reduced density of herbaceous vegetation and shorter woody vegetation (Taillie and Moorman 2019). Habitat suitability models show impoundments fare better than tidal marshes under sea level rise scenarios (Roach and Barrett 2015). Overall, managing marsh habitat for post-fire conditions and impoundments, larger marsh patch sizes, and higher proportions of marsh within a 200-meter buffer may facilitate vegetation migration and increase habitat availability and quality in the face of sea level rise (Roach and Barrett 2015; Taillie and Moorman 2019).

Across their range-wide, high-quality habitat for Black Rail was relatively rare (coastal New Jersey to Texas) and variable at the state level, with the largest proportions of high-quality breeding habitat in North Carolina, South Carolina, and Florida (Stevens and Conway 2021). Average breeding habitat occurred in New Jersey, Delaware, Maryland, and Virginia, with relatively higher quality habitat patches found in New Jersey, Delaware, and Virginia. High-quality breeding habitat was highest on federal and protected lands range-wide, rare in New Jersey and Delaware, and uncommon in Maryland and Virginia (Stevens and Conway 2021). The highest quality patches were rare and unevenly distributed across the range, with 33-42% vulnerable to sea level rise and at increased risk from extreme events (USFWS 2019, Stevens and Conway 2021).

**Indirect effects**

Like other marsh birds, Black Rails face increased negative species trophic interactions due to intra- and interspecies range shifts overlapping habitat use. For example, as marsh habitat availability and quality continue to be threatened by climate-related inundation, the preferred high and freshwater marsh habitat of Eastern Black Rails are being constricted between upland forests and brackish or salt marsh, resulting in increasing competition for food resources with other marsh birds (Roach and Barrett 2015).

**Black Skimmer**

**Population Changes**

Rising sea levels threaten salt marsh island and oceanfront beach nesting habitat for Black Skimmer (Tattoni et al., 2020). In New Jersey, the distribution of nesting Skimmer colonies shifted away from oceanfront beach habitats to salt marsh islands by the 1970s,
presumably in response to increased development and human disturbance (Tattoni et al., 2020). Since 1976, however, Black Skimmer colonies have increasingly shifted back to suboptimal oceanfront beach nesting habitat (Tattoni et al., 2020). The probability that a Black Skimmer nesting colony would be located on salt marsh islands versus oceanfront beaches was 1.71 times higher in 1976 than in 2019 and increased nesting overlap with Least Terns by over 5 times (Tattoni et al., 2020).

Indirect effects

Black Skimmer historically nested along oceanfront beaches in New Jersey but relocated to islands in nearby salt marshes in response to increased development in the 1970s and 1980s (Tattoni et al., 2020). Sea level rise now increasingly threatens these salt marsh island habitat refuges, and skimmers are shifting back to now suboptimal beach habitat where they may compete for nesting habitat with Least Tern and other beach-nesting birds (Tattoni et al., 2020). The recent increase in Black Skimmer colonies on unprotected public beaches has increased the threat of human disturbance, though the major habitat changes in salt marshes may be more impactful than the threat of human disturbance on oceanfront beaches, with Black Skimmers unable to adapt to changing conditions within the salt marshes (Tattoni et al., 2020). The long-term impacts of increased coexistence with Least Terns in Black Skimmer colonies and the potential increase in reliance on conservation management of developed oceanfront beaches with high levels of human disturbance are uncertain (Tattoni et al., 2020).

Blue-winged Warbler

Shifts in Range, and Elevation, or Depth

Distribution models projecting habitat suitability to 2060 suggest varying potential changes in suitable habitat for Blue-winged Warblers depending on which climate change scenarios are considered (Hightower et al., 2022). Under some emission scenarios, Blue-winged Warblers are projected to gain up to 38% total area to their range, yet in other scenario projections, they might lose up to 95% (Hightower et al., 2022). These models also project the center of their range may shift 0.67424° latitude south from the historical center (44.49523° to 43.82099°) or, in others, shift 1.14895° north (to 45.64418° latitude), depending on the
emissions scenario (which range from RCP 2.6 – 8.5; Hightower et al., 2022) which highlights the uncertainty surrounding climate change projections.

**Cerulean Warbler**

*Shifts in Phenology*

Cerulean Warblers did not appear to change their migration or egg-laying dates in concert with the advancement of earlier spring-time warming in Indiana, arriving only 4 days earlier between 1982 and 2019, while spring-time warming was 14 days earlier during the same period (Connare and Islam, 2022).

**Common Tern**

*Shifts in Phenology*

Similar to Roseate Terns, Common Terns are threatened across their Northeast range by changes in prey availability during the critical pre-breeding and post-migration periods (Staudinger et al., 2019; Bratton et al., 2022). Common Tern adult and chick diets are more generalized than other terns that breed in the Gulf of Maine; however, their diets are still reliant on a few prey groups, including sand lance, herring, and hake species (Yakola et al. 2021; Bratton et al., 2022; Legett et al., 2023).

Breeding timing is an important driver of fitness in many bird populations, but identifying specific environmental drivers of seabird breeding phenology is lacking for most populations (Keogan et al., 2022). For seabirds in the North Atlantic (including Common and Roseate Terns), breeding phenology has responded idiosyncratically to local environmental conditions, including sea surface temperatures (Keogan et al., 2022). A lack of evidence for a shared variance in intraspecific breeding phenology across populations and phenotypic plasticity in laying timing in Africa and Europe may be a sign of climate resiliency in this species (Dobson et al., 2017; Keogan et al., 2022).

*Population changes*

In a study conducted outside the Northeast region, the most important environmental predictors for Common Tern nest distribution in the face of sea level rise were habitat type and elevation, with habitat diversity and distance to anthropogenic disturbance as secondary
predictors (Ivajnšič et al., 2017). Projected increases in frequency and intensity of heavy rainfall events and storms are also of concern in affecting nest inundation and failure (Seavey et al., 2011; Ivajnšič et al., 2017).

**Eastern Whip-poor-will**

*Shifts in Phenology*

In Ontario, Eastern Whip-poor-wills seem to be shifting their breeding phenology to become out of sync with peak moth activity, a resource they need to ensure chick survival (English et al., 2017), though the study only covered 3 years (2011-2013), so long-term inferences related to climate change may be hard to make with any certainty. Lower nest success was partially mitigated by the increased renesting after nest failure (English et al., 2017); however, the study only covered a 3-year period (2011-2013), preventing definitive inferences related to climate change.

**Golden Eagle (Eastern population)**

*Shifts in Range, and Elevation, or Depth*

From 1975-2011, the center of the Golden Eagle’s winter range shifted north at a rate of 7.74 km/year, though the study did not test whether climate played a role; however, this range shift was not directly attributed to climate change and could have been the result of other factors (Paprocki et al., 2014).

*Shifts in Phenology*

Warming temperatures have led to shorter residency times on Golden Eagle breeding grounds throughout their North American range; this is in part due to the initiation of their spring migration 10 days later and leaving their breeding ground 20 days earlier during years that were warmer than the decadal average (Maynard et al., 2022).

*Indirect Effects*

Though not in the Northeast, Mexican Chicken Bugs (*Haematosiphon inodorus*) have spread into the western breeding range of the Golden Eagle as far north as Idaho due to climate change, with negative effects on chicks in the nest (Dudek et al., 2021). Climate change may also
lead to lower-quality hunting habitats by providing more areas for prey to hide due to increased leaf cover (Katzner et al., 2023).

**Golden-winged Warbler**

*Shifts in Range, and Elevation, or Depth*

Distribution models projecting habitat suitability to 2060 suggest varying potential changes in suitable habitat for Golden-winged Warblers (Hightower et al., 2022). Under some emission scenarios, Golden-winged Warblers are projected to lose about 10% of total suitable habitat area, yet in other scenarios, projections of suitable habitat loss are more severe, with up to 98% loss while these models also project that the center of their range may shift northwards from the historical center of 42.95637° latitude to between 44.41175° to 48.94493° latitude, depending on the emissions scenario (Hightower et al., 2022).

**Pale-bellied Brant**

*Population Changes*

The population of the U.S. Atlantic Coast population of Pale-bellied Brant is estimated to be stable in the long term (1961-2016), but the short-term population trend (2007-2015) is declining (Smith et al., 2020).

*Indirect Effects*

Sea level rise threatens the salt marsh habitat of migrating and wintering Pale-bellied Brant, which forage on salt marsh plants during high tides when preferred submerged aquatic vegetation and algae are not readily available (Clausen et al., 2013a). In areas where aquatic vegetation is in decline, Pale-bellied Brant and other waterfowl using tidal wetlands and estuaries increasingly use terrestrial habitats for food resources (Clausen et al., 2012, 2013b). Salt marsh vegetation is now a valuable supplemental or alternative food resource during the majority of these birds’ annual cycle and may be an adaptation that supports pre-migration nutrition during spring (Clausen et al., 2013a). Pale-bellied Brant are projected to have decreasing access to important food resources as salt marsh habitat availability is predicted to decline significantly due to sea level rise, warming, higher precipitation, and losses due to coastal squeeze (Clausen et al., 2013a).
**Piping Plover**

*Shifts in Range and Elevation*

Piping Plovers have been relatively well-studied in the context of climate change impacts on coastal environments of the Atlantic Coast breeding population and of nonbreeding birds, yet much of the impact on the Great Lakes and Northern Great Plains breeding populations remains poorly understood (USFWS 2020b).

*Shifts in Phenology*

The mean calendar hatch date for Great Lakes Piping Plovers has shifted four days earlier over the past two decades, possibly due to climate change and warmer spring conditions earlier in the breeding season (Brudney et al., 2013, USFWS 2020b).

*Changes to Morphology or Physiology*

The body mass of Atlantic Coast breeding Piping Plovers varies with latitude and temperature on breeding and wintering grounds, consistent with Bergmann’s Rule and the heat conservation hypothesis (Gibson et al., 2019). Higher body mass and wing length are associated with higher latitudes and cooler temperatures, with variation in wing length most likely due to conditions during the breeding season or migration tradeoffs (Gibson et al., 2019, USFWS 2020b). Shifts in environmental conditions due to climate change may disproportionately affect birds from specific breeding latitudes (Gibson et al., 2019). Without adequate flexibility at the individual level, potential mismatches between environment and phenotype may have population-level impacts with lower persistence of local populations and, ultimately, shifts in their distribution (Gibson et al., 2019).

Environmental factors at breeding sites are the main drivers of trait variation for natal dispersal and female breeding time in Great Lakes Piping Plovers (Saunders and Cuthbert 2014). Both traits have low heritability and evolvability, which may limit long-term adaptive capacity to climate change (Saunders and Cuthbert 2014).

*Population Changes*

The annual survival of Piping Plovers appears to be influenced relatively more by the location of their wintering grounds than their breeding location (Roche et al., 2010; Gibson et al.,
Wintering habitats with significant ecosystem modifications and recreational disturbance had 10% lower survival rates and lower body condition (7% less body mass) than individuals in less disturbed habitats (Gibson et al., 2018). Body condition decline was linked to less time spent foraging and increased stress levels compared to undisturbed habitats (Rutter, 2016; USFWS, 2020b). Overall, Piping Plovers have low adaptive capacity (Saunders and Cuthbert, 2014) and high site fidelity at breeding and wintering sites (Gibson et al., 2017 & 2018; USFWS, 2020b). Projections indicate that Piping Plover populations will lose critical nesting habitat due to the dual pressures of sea level rise and urban development (National Wildlife Federation and Manomet Center for Conservation Sciences, 2014; Seavey et al., 2011; Galbraith et al., 2014; USFWS 2020b). One study projected a more than 50% loss of Piping Plover specialized habitat throughout their lifecycle due to rising sea levels (Galbraith et al., 2014).

Nesting Piping Plovers have been observed to utilize newly suitable habitats following coastal storms or flooding events for several years, leading to local population increases (Cohen et al., 2009; Schupp et al., 2013; Bourque et al., 2015; Maslo et al., 2019; Robinson et al., 2019; Walker et al., 2019; Zeigler et al., 2019; USFWS 2020b). Overwash sand deposits that increase coastal elevation can counteract the vulnerability of nests to increased flooding and sea level rise (Sweet et al., 2017). Protection and maintenance of new overwash habitats and opportunities for new habitats to form during natural disturbance events can benefit Piping Plovers by increasing carrying capacity and population growth (Hecht and Melvin, 2009; Zeigler et al., 2017b; Robinson et al., 2019; Stantial et al., 2021). Sea level rise also may impact groundwater systems on barrier islands (Masterson et al., 2013), which could impact the character, amount, and distribution of valuable moist sediment substrates for foraging Piping Plovers or the composition of barrier island vegetation (USFWS, 2020b).

Indirect effects

American Oystercatchers are known to have aggressive encounters with Piping Plover, which is suspected to contribute to nest failure (Hogan et al., 2018; Robinson et al., 2019). Aggression between Piping Plover and Wilson’s Plover (*Charadrius wilsonia*) is also hypothesized to influence territoriality behaviors and increased spacing of beach-nesting habitat of the two shorebirds in Virginia (Bergstrom and Terwilliger, 1987; Robinson et al., 2019). As
coastal squeeze decreases habitat availability, these behavioral and competitive interspecific interactions among RSGCN bird species may become amplified and have the potential to reduce nest success, chick survival, and overall productivity.

**Purple Sandpiper**

*Population Changes*

Purple Sandpipers specialize in foraging on exposed rocky shorelines and roost during winter. Rocky shoreline habitat, either natural or engineered, does not migrate inland due to rising sea levels. Therefore, Purple Sandpipers are expected to lose area and/or time available for foraging as tidal heights change under future climate projections (Nightingale et al., 2018).

**Roseate Tern**

*Shifts in Phenology*

The onset of spring in the Gulf of Maine has occurred earlier in recent years, which can impact prey phenology during the critical period for the breeding success of Roseate Terns (Staudinger et al., 2019; Bratton et al., 2022). Roseate Terns are highly specialized foragers and rely heavily on sand lances (*Ammodytes* sp.) during the breeding season; this trait paired with the fact that the vast majority of the Roseate Tern population nests on three islands off of Connecticut and Massachusetts outs them at high risk to changes in prey availability and extreme events (Yakola et al., 2021; USFWS, 2020c; Staudinger et al., 2020). Changes in the availability of primary prey resources have been linked to decreased chick survival and nest failure outside the Northeast region (Catry et al., 2013) and are of high concern in the Northeast. Factors influencing the distribution and abundance of sand lance are poorly known because they are not well sampled in fisheries surveys (Staudinger et al., 2020). While there is some evidence for adaptation to local conditions on breeding grounds (Legett et al., 2023), additional research and monitoring of Roseate Tern diets paired with surveys of sand lance distribution are critical for determining whether Roseate Terns can switch to novel prey if sand lance becomes less available in the future.
**Rufa Red Knot**

*Shifts in Range and Elevation*

Red Knots and other birds with low-temperature index values exhibit more negative population trends associated with higher temperatures (Anderson et al., 2023). Population declines are compounded by declines in habitat availability on their Arctic breeding grounds (Anderson et al., 2023). Indeed, the USFWS has identified four large wintering regions of Rufa Red Knots (USFWS 2021a) with observed contractions of 75% on the southern wintering region in southern South America during the 2000s and a lack of recovery as of 2021 (USFWS 2021a).

*Shifts in Phenology*

Red Knots are threatened by climate-induced phenological mismatches in food resources during breeding (Lameris et al., 2022) and migration (Tucker et al., 2019). Chicks of four subspecies of Red Knot, not including *Calidris canutus rufa*, experienced growth reductions when they hatched after the seasonal peak in arthropod abundance and growth (Lameris et al., 2022).

Rufa Red Knots that winter along the southeastern U.S. coast feed on intertidal invertebrates, including mollusks, insects, and annelids (Manrique 2020) (USFWS 2021a). However, 50-80% of the entire population of Red Knots specialize in Horseshoe Crab eggs during their stop-over period in the Delaware Bay estuary (Tucker et al., 2019, Manrique 2020, USFWS 2020a & 2021a). The timing of availability of Horseshoe Crab eggs is linked to water temperatures, with crabs spawning later in cooler years, and was associated with delayed and faster gains in mass for Rufa Red Knots (Tucker et al., 2019). Conversely, Ruddy Turnstones (*Arenaria interpres*), a more generalist species, were unaffected by Horseshoe Crab phenology changes (Tucker et al., 2019).

*Changes to Morphology or Physiology*

Red Knots born in warmer years with early snowmelt had smaller body sizes and shorter beaks than birds born in colder years, possibly due to malnutrition in earlier life stages (van Gils et al., 2016). Shorter-billed birds had lower survival rates due to difficulties foraging for their preferred mollusk prey, which are deeply buried and were forced to switch to relatively shallow-buried seagrass rhizomes (van Gils et al., 2016).
Changes in Population

The short-term (2001-2016) and long-term (1974-2016) population trends of the Rufa Red Knot are negative (Smith et al., 2020). All six subspecies of Red Knot are experiencing population declines (Manrique, 2020), and overall populations remain depleted (USFWS, 2021a), although population status and trends for three of the four wintering regions remain uncertain (USFWS, 2021b). The number of Rufa Red Knots stopping at Delaware Bay during peak migration has recently declined (USFWS, 2021a). Population declines continue despite recent stability in Horseshoe Crab populations at migratory stopovers (USFWS, 2021a), suggesting that factors other than food at this location influence survival (Manrique, 2020). One possible explanation for overall population declines is linked to declining snow cover, drier conditions, and mismatches in prey resource availability on their Arctic breeding grounds (Meltofte, 2007; Smith et al., 2010; Manrique, 2020). These changes have reduced weight gain and increased starvation of juveniles who cannot survive fall migration to their wintering grounds (Manrique 2020).

Red Knots are threatened by extreme weather, precipitation, and storm intensity or frequency increases at key migration stopover locations (USFWS 2014, 2020a, 2021a). The USFWS identified climate change threats of increasing frequency and severity of migration asynchronies, rising sea levels, and changing Arctic ecosystems in the suite of primary justifications for listing the Rufa Red Knot as federally threatened (USFWS 2020a, 2021a). Sea level rise projections decrease the future availability and stability of wide tidal flats, channel inlets, and barrier islands, which are optimal non-breeding habitats for Red Knots (USFWS 2021a).

Indirect effects

Major fall migration stopover locations in the Gulf of Maine and Gulf of St. Lawrence warmed by more than 2°C when comparing 1880 to 1899 with 2014 to 2018 (Mooney and Muyskens 2018). Warming ocean temperatures, ocean acidification, and hypoxia impacts of climate change resulted in negative and complex responses by their marine bivalve prey in the North Atlantic (Stevens and Gobler 2018, USFWS 2021a, b).
Ruffed Grouse

*Shifts in Range, and Elevation, or Depth*

Species distribution models, combined with climate change scenario projections (RCP 4.5) and phylogeographic information, projected that the Ruffed Grouse’s range will shift northerly but expand to include more area than it has experienced by 2050 and 2070 (Perktaş, 2020).

Rusty Blackbird

*Shifts in Range, and Elevation, or Depth*

Models suggest that despite the highest emission scenario (RCP 8.5), 43% of Nova Scotia will likely remain suitable for Rusty Blackbirds, suggesting that this region may serve as climate refugia for this species in the future (Bale et al., 2020).

Saltmarsh Sparrow

*Changes in Population*

Due to increasing sea levels and flooding, Saltmarsh Sparrow populations declined throughout their breeding range, from Maine to Virginia, at a rate of approximately 5% per year from 2000 to 2013 (Shriver et al., 2015). This trend will likely continue regardless of which climate change scenario is used to project the effect of sea level rise on the breeding success of Saltmarsh Sparrows; estimates suggest that they may go extinct as soon as 2035 but almost certainly by 2060 (Field et al., 2016), which is similar to estimates in New Jersey, which projects they will become locally extinct before 2050 (Roberts et al., 2019). While females do show some plasticity in selecting higher nest sites after nest failure due to flooding, it is unlikely that this will be enough to cope with the projected increases in flooding associated with sea level rise projections (Benvenuti et al., 2018).

Summary

Overall information is still needed to identify environmental drivers of breeding phenology and the degree of plasticity or adaptive capacity to respond or adapt to changing conditions, including migration cues and vulnerability to phenological mismatches. Predictions or forecasts for species-specific habitat availability and suitability for breeding, foraging,
migration, and wintering would be helpful to characterize future risks to habitat squeeze and shifting ranges. Changes in food availability for diet specialists and their plasticity or adaptive capacity to shift to alternative food resources, with subsequent implications for nesting success and productivity, are needed for many colonial breeding species. Tracking individual patterns in habitat use in response to sea level rise and marsh migration can increase understanding of long-term population trends in the region, especially for rare and cryptic species (Roach and Barrett, 2015; Taillie and Moorman, 2019).

Much of the avian research has focused on marine and coastal birds, but there are many research needs for land birds. Evaluating whether the timing of annual surveys with fixed dates, such as the Christmas Bird Count (CBC), does not inadvertently mistake population declines when range and/or phenology shifts cause population distribution changes would be useful for many migratory species. Understanding how changes in food availability affect interspecific competition, especially as species shift their phenology and range, is needed (Schummer et al., 2020; Thurber et al., 2021). Continued and expanded monitoring surveys of species that link population trends in the Northeast with their wintering ranges can be used to test hypotheses of whether the ranges are contracting or colonizing new areas and if there is sufficient foraging habitat carrying capacity to support shifting populations (Meehan et al., 2021).

C) DIADROMOUS FISH

Table 2.5. RSGCN diadromous fish species and number of scientific articles cited in this report for each species.

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<th>Common Name</th>
<th>Scientific Name</th>
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<td>American Eel</td>
<td><em>Anguilla rostrata</em></td>
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<td>American Shad</td>
<td><em>Alosa sapidissima</em></td>
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**Figure 2.4.** Number of scientific articles for all RSGCN diadromous fish by biological response type.

**Overview**

Warmer temperatures, higher salinity, lower dissolved oxygen, increasing ocean acidification, and changing water currents have and will continue to, affect already declining populations of diadromous fishes (Kerr et al., 2009) from shifts in range, depth, or elevation (e.g., Nye et al., 2009); changes to phenology (e.g., Staudinger et al., 2019; Otero et al., 2014);
and increased mortality risk from hypoxia in warming ocean water (e.g., Secor and Gunderson, 1998). Secondary effects of climate change may include decreased prey availability (e.g., in American Shad nurseries; Crecco and Savoy, 1984) and increased disease risk (e.g., sea lice [Lepeophtheirus salmonis] in the Gulf of Maine; Bricknell et al., 2021). Climate change has impacted recruitment in some diadromous fish, including Alewife (Tommasi et al., 2015) and American Eels (Miller and Casselman, 2014).

“River Herring” (Alewife and Blueback Herring)

Shifts in Range, Elevation, or Depth

Suitable thermal habitats for both Alewife and Blueback Herring (hereafter “river herring” when these two species are considered together) will likely decline over time (Lynch et al., 2015). These species may prosper in Canadian waters while becoming sparse along the U.S. Atlantic Coast (Reist et al., 2006; Lynch et al., 2015). However, the degree to which these two species can shift to find cooler waters may be limited by their behavior of returning to their natal spawning sites (Lynch et al., 2015), which is already hampered by dams and other riverine and coastal barriers to movement (Alcott et al., 2021). Evidence suggests that the Alewife’s center of biomass is shifting considerably (5.47 km/year north) and that they are shifting to deeper, more offshore habitats in the ocean (1.15 m/year; Nye et al., 2009). As Alewives shift into deeper marine waters, they must migrate further distances to spawn if they return to their freshwater natal spawning grounds (Hare et al., 2016).

The spring range of Alewife shifted 1.72 degrees (191.76 km) north and contracted by 0.35 degrees between the periods of 1974-1977 to 2019-2022; conversely, during fall, Alewife shifted 0.46 degrees (51.25 km) south and expanded its range by 0.09 degrees (11.03 km) between the periods of 1974-1976 to 2019-2022. Alewife have shifted their depth deeper by 79.9 meters during spring and 37.5 meters during fall over similar periods (NOAA Fisheries, 2022). Blueback Herring shifted 2.5 degrees (277.81 km) north and expanded its range by 0.42 degrees (47.42 km) during spring between the periods of 1974-1977 to 2019-2022. Over similar periods, Blueback Herring shifted to deeper waters by 66.1 meters (NOAA Fisheries, 2022).

Alewife are projected to experience a significant loss of relative biomass during fall by 2050 across the Northeast continental shelf; slightly higher decreases are projected during spring and fall in the Southern New England – Mid-Atlantic regions, while slight increases are projected in
the Gulf of Maine during spring under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is high certainty in projections that Alewife will shift the centroid of their range across the Northeast continental shelf by 149.9 km under RCP 2.6 and medium uncertainty they will shift 141.6 km under RCP 8.5 by the end of the century (2081-2100), with projected losses of available thermal habitat of -24.9% under a RCP 2.6 and -71.5% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018). There is high certainty that Blueback Herring will shift the centroid of their range across the Northeast continental shelf by 204.7 km under RCP 2.6 and 400.2 km under RCP 8.5 by the end of the century (2081-2100), with projected losses of available thermal habitat of -2.3% under RCP 2.6 and -66.5% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

Shifts in Phenology

In Southern New England, adult Alewife started their spawning run 13 days earlier in the 2000s than they did in the 1970s (Ellis and Vokoun 2009) and 21.6 days earlier in the mid-coastal region of Maine between 1983 and 2001 (Huntington et al., 2003). In the southern part of their range, adult Alewife in North Carolina shifted the timing of spawning; between the 1970s and 2010s, spring warming was the best explanation for a shift in the arrival time on their spawning grounds to 16 days earlier and their departure to 27 days earlier, shortening the overall spawning duration period by 11 days total (Lombardo et al., 2019). Similarly, in North Carolina, adult Blueback Herring shifted their arrival time to five days earlier and left 23 days earlier, shorting their spawning period duration by 18 days from the 1980s to the 2010s (Lombardo et al., 2019).

While many studies have shown phenological shifts to earlier in the year, adult Alewife immigrated into freshwater during spring on a fairly consistent schedule across nine sites in Massachusetts for the past 8-28 years; however, a few sites exhibited shifts to later in the year for both run initiation and peak migration timing (Dalton et al., 2022), a finding similar to another set 20 lakes in Massachusetts (Rosset et al., 2017). Phenological responses in this system were thought to be complicated by other factors, including changes in population size and barriers to passage (Alcott et al., 2021) and/or lunar phases (Rossett et al., 2017); it is also possible that climate thresholds that would considerably change the timing and patterns of Massachusetts Alewife spawning runs have yet to be crossed (Staudinger et al., 2019; Legett et al., 2021; Dalton et al., 2022).
With some Alewife subpopulations shifting earlier, some not changing, and others shifting later, the lack of a consistent pattern in river herring phenology, local conditions, and adaptations likely play an important role in determining the degree to which river herring shift as the effects of climate change become more pronounced (Legett et al., 2021; Dalton et al., 2022). The wide variability across sub-populations indicates that local and site-specific conditions are important in designing effective management actions.

Changes in Morphology or Physiology

Warming generally results in smaller aquatic organisms (Staudinger et al., 2021). Alewife have been decreasing in body size, both in marine (Lynch et al., 2015) and freshwater habitats (Palkovacs et al., 2013); however, the results were inconclusive whether the driving factor was, in fact, climate or other factors, including fishing pressure (Lynch et al., 2015). Under experimental conditions, juvenile Alewife exposed to higher temperatures or reduced food availability were smaller and had less fat than those exposed to lower temps and more food, with the results amplified with both treatments (i.e., higher temperatures and less food; Guo et al., 2021). As a potential consequence of decreased size, smaller female Alewife reproduced at lower rates in a whole lake experiment in Massachusetts than larger individuals (Marjadi et al., 2019). While Blueback Herring collected in Connecticut and exposed to higher temperatures had more fat than those exposed to lower temperatures, they were smaller —— similar to Alewife (Guo et al., 2022).

Changes in Population

In a laboratory experiment, fewer juvenile Blueback Herring survived when temperatures were 33°C, suggesting that as ambient temperatures grow warmer, fewer individuals may survive, and those that do will likely be smaller (Guo et al., 2022). Other diadromous fish show similar patterns. Marjadi et al. (2019) also found that smaller female Alewives had lower reproductive rates than larger females. However, models comparing various management actions on future river herring populations suggest that increasing the connection between rivers and ocean (e.g., dam removal; Song et al., 2019), along with reduced fishing pressure, could lead to substantially increased biomass for these species in the Gulf of Maine, which could lead to overall community resilience to climate change (Dias et al., 2022).
American Eel

Shifts in Range, Elevation, or Depth

The latitudinal range of American Eel in the Northeast is unlikely to be impacted since the Northeast is in the northern part of a range that extends south to the Caribbean (Hare et al., 2016).

Shifts in Phenology

The American Eel fishery (for elvers) on the East River (Nova Scotia) started ~ 22 days earlier in 2018 than it did in 1996 (Jessup 2020). Moreover, each increase of 1°C of the sea surface temperature on the coast leads to a 9-day earlier start for American Eel fisheries in Nova Scotia, ranging from 22 to 34-day average earlier start time between 1996 and 2020 (Jessup 2021). It is important to note when using the timing of a fishery rather than the directly measured biological (species) response that while fisheries can be a good indirect measure of changes in phenology, other social, economic, and access factors can influence when fisheries open, which may not perfectly align with the biology of the eels.

Atlantic Salmon (Gulf of Maine Population)

Shifts in Phenology

Atlantic Salmon smolts started emigration from freshwater to the sea 2.5 days earlier per decade between 1960 and 2012, including in Connecticut and Quebec, closely tracking changes in the temperature of rivers, oceans, and the air (Otero et al., 2013). In rivers found in Newfoundland and Labrador, a warming climate has contributed to adult Atlantic Salmon returning to their spawning ground 12 days earlier from 1978 to 2012 on average (though some areas saw salmon return as much as 21 days earlier), leading to a shorter spawning period (Dempson et al., 2016). Similarly, due to warming water in the Connecticut River, Atlantic Salmon started their spawning migration ten days earlier from 1978 to 2000 (Juanes et al., 2004); while, not in the northwest Atlantic Ocean, Atlantic Salmon smolt migration shifted 10 days earlier from 2000 to 2014 in the northern Baltic Sea corresponding to warmer May air temperatures (Jokikoko et al., 2016).
Changes in Morphology or Physiology

Some of the strongest evidence for Atlantic Salmon body size changes is outside the Northeastern U.S. region, in the northern Baltic Sea. Studies found over 14 years, two-year-old Atlantic Salmon smolts were 8 mm smaller, and three–year–olds were 11 mm smaller (mean length of 137 mm to 129 mm and 150 to 139 mm, respectively), though the authors could not resolve the mechanism that led to smaller size-at-age, though density-dependence or decreasing population numbers were not significantly influential (Jokikokko et al., 2016). In an experimental study, exposing fish to warmer water temperatures prevented sexual maturation in both male and female Atlantic Salmon by blocking ovulation in females and milt production in males, while more eggs survived to the eyed stage of development in females exposed to cooler temperatures (Vikingstad et al., 2015). Since smaller smolts are less likely to survive in the open ocean (Kallio-Nyberg et al., 2011), if the trend of salmon smolt getting smaller is combined with inhibited sexual maturation and lower egg survivorship during development with warming waters, Atlantic Salmon, especially in rapidly warming parts of their range, will likely to see a continued decline in population size on top of the precipitous decline they have already experienced throughout the Northeast. Interestingly, these trends of individuals decreasing in size and decreased reproductive success match archeological evidence of historical responses of Atlantic Salmon to warming temperatures starting ~15,000 years ago (Blanchet and Dubut 2012). At temperatures above 21°C, thermal stress changes juvenile Atlantic Salmon behavior (Cunjack et al., 1993), and they also stop feeding (Elliot 1991). Changes in the Nova Scotia Coastal Current affected post-smolt Atlantic Salmon migration in the Gulf of Maine, leading to smaller individuals arriving at their spawning grounds (Byron et al., 2014).

Changes in Population

Atlantic Salmon parr gravitated to cooler water microhabitats in streams and rivers and lingered thereafter high-heat events (Corey et al., 2023), which suggests that in addition to actions like dam removal, identifying and protecting these cool refugia within river systems will be important to help diadromous fish persevere into the future (Corey et al., 2023; Wilbur et al., 2020). To aid in identifying cooler refugia, researchers are developing aquatic tools and models for managers (e.g., Saadi et al., 2021; Alidi et al., 2022). In addition to the conditions diadromous fish encounter during the freshwater phase of their life history, warming in the
Labrador Sea has reduced the available thermal habitat where Atlantic Salmon overwinter and feed (Friedland et al., 2013). Though warming marine habitat for Atlantic Salmon contributes to larger-sized adults, the reproductive benefit of larger adults returning to their spawning grounds will likely be canceled out by the decreasing adult recruitment associated with warmer seas (Friedland and Todd, 2012). While decreased suitable habitat has directly contributed to population declines, the indirect trophic effects of shifts in primary producers and forage fish communities may be more responsible for losses in production and biomass (Mills et al., 2013).

**Indirect Effects**

Warming temperatures in the Gulf of Maine may lead to more virulent Salmon Lice (*Lepeophtheirus salmonis*), decreased growth, and increased mortality of Atlantic Salmon (Bricknell et al., 2020). Since cooler water temperatures reduce Salmon Lice survival and success (Ugelvik et al., 2022), climate change and warmer water temperatures will likely lead to higher infection rates and faster reproduction of the lice (Sandvik et al., 2021).

**American Shad**

**Shifts in Range, Elevation, or Depth**

Between the periods of 1974-1977 to 2019-2022, American Shad shifted 1.73 degrees (192.84 km) north and contracted their spring range by 0.38 degrees (42.02 km) across the Northeast continental shelf; they also shifted 54.2 meters to deeper waters from 1974-1977 to 2019-2022 (NOAA Fisheries, 2022).

American Shad are projected to experience a significant loss of relative biomass during fall by 2050 across the Northeast continental shelf, with higher decreases projected in the Southern New England – Mid-Atlantic regions and slight increases during spring in the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is lower certainty in the projections that American Shad will shift the centroid of their range across the Northeast continental shelf by 65.0 km under RCP 2.6 and medium uncertainty they will shift 337.6 km under RCP 8.5 by the end of the century (2081-2100), with projected changes of available thermal habitat of 20.6% under a RCP 2.6 and -19.9% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).
**Changes in Phenology**

Models suggest that American Shad will likely follow a similar pattern of migrating and spawning earlier as temperatures increase in the Connecticut and York River watersheds (Kerr et al., 2009), much like American Shad populations in the western U.S., in the Columbia River watershed, which saw them migrating five weeks earlier from 1949 and 1993 (Quinn and Adams 1996).

**Atlantic Sturgeon**

*Changes in Population*

Changes in precipitation patterns and increased temperatures will likely impact river hydrology (e.g., streamflow and water temperature), leading to reduced productivity and limited juvenile growth in many diadromous species, especially in the southern part of their range (Overton et al., 2012, Tommasi et al., 2015, Hare et al., 2016). For example, model predictions suggest that even a 1°C temperature increase led to a 65% decrease in Atlantic Sturgeon productivity in the Chesapeake Bay (Niklitschek and Secor 2005). In a laboratory experiment, higher water temperature increased juvenile Atlantic Sturgeon mortality due to hypoxia (Secor and Gunderson 1998), and higher water temperature and salinity led to a decrease in body mass, swelling of red blood cells, and mortality in Shortnose Sturgeon (Downie et al., 2018)

**Summary**

Overall, because of the complex early life history, migration, and use of multiple habitats (freshwater and marine) of RSGCN diadromous fish, all but one are considered “Very High” climate change vulnerability risk; American Eels are listed as “High” (Hare et al., 2016 and see Chapter 3). Further, the cumulative responses of shifts towards smaller body size and lower reproductive potential suggest that climate change may lead to a cycle of population decline in the warmer parts of diadromous fishes’ ranges. Contraction of phenophases, as demonstrated in studies (e.g., Atlantic Salmon [Dempson et al., 2016]; Alewife and Blueback Herring [Lombardo et al., 2019]), can increase the likelihood of risk to extreme events from spring storms (e.g., Nor’eaters) (Lynch et al., 2015) or anthropogenic actions like water releases from hydropower facilities.
D) MARINE & COASTAL FISH

Table 2.6. RSGCN marine and coastal fish species and number of scientific articles cited in this report for each species.

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**Figure 2.5.** Number of scientific articles for all RSGCN marine and coastal fish and elasmobranch species by biological response type.
Overview

Many studies provide strong evidence that marine fish and elasmobranch species respond to climate change through geographical range (Nye et al., 2009) and phenology shifts (Staudinger et al., 2019; Langan et al., 2021). Studies of range shifts generally use long-term federal datasets from spring and fall bottom trawl (NOAA Fisheries, 2022) and ichthyoplankton Ecosystem Monitoring (EcoMon) (Walsh et al., 2015) surveys of the continental shelf or by state and private surveys of regional bays and estuaries (Morson et al., 2019, Snyder et al., 2019; Langan et al., 2021). These extensive monitoring programs have shown that centers of population biomass for most marine fish species are shifting poleward towards higher latitudes and to greater depths at rates that exceed terrestrial species (Lenoir et al., 2020). These general trends are projected to continue under future climate scenarios of modeled habitat abundance (Kleisner et al., 2017), suitable thermal habitat (Morley et al., 2018), and relative population biomass (Allyn et al., 2020).

An analysis of 42 species of marine fish and invertebrates (RSGCN and non-RSGCN species) found in continental shelf waters of the Northeast region between 1974 and 2022, showed the community has shifted on average 0.75 degrees (83.83 km) north during spring, and 0.61 degrees (68.27 km) north during fall. The community also experienced a change in average depth of 22.4 meters and 8.6 meters deeper in spring and fall, respectively (NOAA Fisheries, 2022). Similar to patterns observed at the community scale and for non-RSGCN species, the direction, rate, and magnitude of range shifts varies widely among RSGCN marine fishes and elasmobranchs sampled during this period; responses in range changes span -0.51 km southward in Winter Skate, to 2.25 km northward in Black Sea Bass during spring, and -0.21 km northward in Atlantic Herring to 2.05 km northward in Black Sea Bass during fall. Depth range shifts spanned from -61.08 meters deeper in Atlantic Herring to 12.41 km shallower in Atlantic Cod in spring and -50.88 meters deeper in Atlantic Herring to 31.20 meters shallower in Atlantic Halibut during fall (NOAA Fisheries, 2022).

Shifts in phenology are more difficult to track compared to range shifts in aquatic species because observations need repeated observations within a narrow and set area over multiple decades (Staudinger et al., 2019). Several multi-species analyses have shown phenological shifts in the occurrence of larval phases of demersal groundfishes (Walsh et al., 2015), growth and productivity of population biomass (Henderson et al., 2017), as well as seasonal migration
patterns (both ingress and egress) and residence times in estuaries systems such as Narragansett Bay, Rhode Island (Langan et al., 2021).

As species make distributional changes in range and phenology, primarily in response to warming sea surface and bottom temperatures, entire assemblages of marine species are reshuffling and reorganizing across Northeast coastal ecosystems (Weiskopf et al., 2020; Staudinger et al., 2021). Examples of changes in community structure through altered species diversity, population size, and density have been documented across the Northeast continental shelf (Friedland et al., 2020) and in estuaries including Little Egg Inlet, New Jersey (Morson et al., 2019), Narragansett Bay, Rhode Island (Langan et al., 2021), and Long Island Sound off of New York and Connecticut (Snyder et al., 2019). Ensembles of warm-water-associated species that are moving north into temperate habitats are generally replacing cold-water-associated species, which are moving to deeper depths and higher latitudes in search of cold-water refugia (Friedland et al., 2020; Pershing et al., 2021).

On a global scale, warming has led to a decrease in body size (Sheridan and Bickford 2011) but this varies widely by species and local conditions. The indirect effects of morphological and behavioral changes combined with shifting distributions are changing predator-prey and competitive relationships among associated marine fish species and resulting in novel trophic interactions (Staudinger et al., 2021). Most models that project range shifts currently do not account for species interactions (Pearson and Dawson, 2003; MacPherson et al., 2021); this is an area of active research as changes in trophic relationships are hypothesized to be more important for species survival under future conditions than environmental conditions (Louthan et al., 2015, Staniczenko et al., 2017). In addition, the combined stress from warming ocean temperatures, decreased dissolved oxygen, and ocean acidification is increasing the prevalence and vulnerability to disease and predation.

**Atlantic Cod**

*Shifts in Range or Depth*

Between the periods of 1974-1977 and 2019-2022, the spring range of juvenile and adult Atlantic Cod shifted 0.32 degrees (36.56 km) north and contracted by 1.13 degrees (125.08 km) across the Northeast continental shelf; conversely, during the fall season, Atlantic Cod shifted 0.15 degrees (16.74 km) south and contracted its range by 1.12 degrees (124.7 km) between the
periods of 1974-1976 to 2019-2022. Atlantic Cod have also shifted 0.5 meters into shallower waters during spring and 12.4 meters shallower during fall over the two periods (NOAA Fisheries, 2022). Larval Atlantic Cod shifted their distribution northward and to deeper waters between the periods of 1977-1987 and 1999-2008 in the Southern New England to Gulf of Maine regions (Walsh et al., 2015). Over the same time period, adult and larval life phases of Atlantic Cod exhibited synchronous distributional shifts to deeper waters during spring across the Northeast continental shelf region (Walsh et al., 2015).

While Atlantic Cod increased in number off of Rhode Island in Southern New England waters between 2000 and 2018 (Langan et al. 2020), they are projected to experience significant losses of relative biomass during spring and fall by 2050 across the Northeast continental shelf with larger declines in the Southern New England – Mid-Atlantic regions compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). This loss of biomass (and range) along the Scotian Shelf to the southernmost parts of the range is likely due to increasing water temperatures and declining oxygen levels (Deutsch et al., 2015), though modeling suggests that warming temperature might be more limiting than oxygen levels except in the deepest parts of the Atlantic Cod’s range (Brennan et al. 2016). There is low certainty that Atlantic Cod is projected to shift the centroid of their range along the coast of Eastern Canada by 111.6 km under RCP 2.6 and medium uncertainty they will shift 428.2 km under RCP 8.5 by the end of the century (2081-2100), with projected losses of available thermal habitat of -1.7% under a RCP 2.6 and -34.0% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

Mechanistic distribution models based on climate change projections and various habitat suitability for Atlantic Cod at different phases of their life history suggest a general northern shift; however, habitat shifts for different life stages vary in projections (Cote et al., 2021). Specifically, the best habitat for egg survival may expand by 153% by 2010, but the habitat for post-hatching growth may decrease by 74% (Cote et al., 2021). Another model that takes into account projected changes in temperature and hypoxic conditions in the Gulf of St. Lawrence projects that Atlantic Cod will lose ~5% of biomass by 2065 due to losing habitat in deeper areas of the Gulf as they become more hypoxic and will remain in shallower, more oxygenated waters (Stortini et al. 2016).
Changes in Morphology or Physiology

In an experimental study, larval Atlantic Cod were exposed to ocean acidification levels projected in the year 2100 (1179 µatm CO$_2$), and the larval mortality rates were doubled compared to those exposed to control levels of acidification (503 µatm CO$_2$; Mittermayer et al., 2019). Similarly, another experimental study exposing Atlantic Cod larvae to the same projected ocean acidification levels (1179 µatm CO$_2$) and varying levels of food limitations found that under high acidification levels (but limited in food), larvae were larger at 36 days and showed organ defects and smaller gills, suggesting trade-offs between growth and organ development that can have serious ramifications for the larvae as they continue to develop (Stiasny et al., 2019). In another study, larvae hatched from parents acclimated to high acidification levels (1100 µatm CO$_2$) still had much higher levels of mortality than those raised in a controlled environment, but some of the mortality was mitigated by high food availability, demonstrating that acclimation may help survival in high acidification levels, but only in ideal circumstances (Stiasny et al., 2018). Eggs exposed to a control and elevated acidification levels (400 and 1100 µatm CO$_2$, respectively) and temperature at the upper end of their thermal tolerance (12°C) were significantly less likely to hatch, and those that did were much more likely to be deformed (Dahlke et al., 2017).

Population Changes

Between the years 1982 and 2013, the Gulf of Maine warmed faster than 99.1% of the rest of the world’s oceans (Pershing et al. 2015), which is correlated with an increase of mortality of age 4 Atlantic Cod (Pershing et al. 2015), which is the year they start preparing for spawning and switching between benthic to pelagic prey (Sherwood et al., 2007), which makes them less efficient predators (Pershing et al. 2015), something that would be further worsened by the increased metabolic costs of rising water temperatures (Deutsch et al., 2015). In the Gulf of St. Lawrence, models suggest that warming waters make it likely that the Atlantic Cod population may only slightly recover their population by 2040 with severe limitations on fishing and predation by Harp Seals (Pagophilus groenlandicus; Bousquet et al., 2014).
Indirect Effects

Warming temperatures will likely lead to changing predator-prey dynamics, as range overlap of Atlantic Cod and their major prey (e.g., Sandlance *Ammodytes dubius* and Atlantic Herring) off of the Northeast U.S. decreased from 1968 to 2014 and is projected to further decrease by as much as 50% in the next ~80 years by based on the RCP 8.5 emissions scenario, severely impacting the food web and top-down control of Cod in the region (Seldon et al., 2018).

Atlantic Halibut

Shifts in Range or Depth

Between the periods of 1974-1977 to 2019-2022, the spring range of Atlantic Halibut across the Northeast continental shelf region has shifted 0.55 degrees (62.18 km) north, and their range has expanded by 0.21 degrees (23.92 km); conversely during the fall season, their fall range has moved 0.18 degrees (20.26 km) south and contracted by 1.02 degrees (112.8 km) between the periods of 1974-1976 to 2019-2022. Atlantic Halibut has shifted to shallower depths in spring and fall by 5 meters and 31.2 meters, respectively (NOAA Fisheries, 2022).

Atlantic Halibut are projected to experience significant losses of relative biomass during spring and fall by 2050 across the Northeast continental shelf under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Atlantic Halibut will shift the centroid of their range along the coast of Eastern Canada by 122.1 km under RCP 2.6 and medium uncertainty they will shift 440.9 km under RCP 8.5 by the end of the century (2081-2100), with projected changes of available thermal habitat of 5.5% under a RCP 2.6 and -18.6% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

Changes in Morphology or Physiology

Multiple studies have shown that Atlantic Halibut is sensitive to the effects of ocean acidification and warming water temperatures. Health effects include reduced growth rates (Grans et al., 2014), metabolism increases and apoptosis signaling in gills and blood plasma (Bresolin et al., 2014). Juvenile Atlantic Halibut had increased immune system responses under varying conditions (5-18 °C and pH of 8.0 and 7.6) that caused increased oxidative damage (Almroth et al., 2019) and may impact their ability to acclimatize (de Souza et al., 2016).
Atlantic Herring

Shifts in Range or Depth

Between the periods of 1974-1977 to 2019-2022, juvenile and adult Atlantic Herring shifted their spring range across the Northeast continental shelf region by 1.24 degrees (138.7 km) northwards, and their range has contracted by 0.01 degrees (2.16 km). Conversely, during fall, their range shifted 0.21 degrees (23.43 km) south and expanded by 0.91 degrees (101.07 km) between 1974-1976 to 2019-2022. Atlantic Herring have shifted to deeper waters during spring and fall by 61 meters and 50.8 meters between 1974-1977 to 2019-2022 and 1974-1976 to 2019-2022, respectively (NOAA Fisheries, 2022). Larval Atlantic Herring shifted their distribution further offshore between the periods of 1977-1987 and 1999-2008 in the Georges Bank to Gulf of Maine regions (Walsh et al., 2015). Over the same periods, adult and larval life phases of Atlantic Herring exhibited synchronous distributional shifts to offshore waters during spring across the Northeast continental shelf region (Walsh et al., 2015).

Atlantic Herring are projected to experience a significant loss of relative biomass during spring and fall by 2050 across the Northeast continental shelf with larger declines in the Southern New England – Mid-Atlantic compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Atlantic Herring will shift the centroid of their range across the Northeast continental shelf by 94.3 km under RCP 2.6 and 380.0 km under RCP 8.5 by the end of the century (2081-2100), with projected changes of available thermal habitat of 1.7% under RCP 2.6 and -12.6% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

Changes in Morphology or Physiology

Atlantic Herring contracted their winter residence time in Narragansett Bay, Rhode Island by 16 days between 1959 and 2016; however, the change was not statistically significant and could not be definitely attributed to water temperatures, possibly due to insufficient sampling (Langan et al., 2021). In the Baltic Sea, Atlantic Herring started developing from smaller (<10 mm) to larger (>15 mm) larvae 7.7 days per decade earlier between 1974 and 1996, coinciding with warming temperatures (Weigel et al., 2021).
Shifts in Range or Depth

In places where Atlantic Herring lay their eggs deep in the water column, which typically have low concentrations of oxygen but high levels of CO₂, their spawning habitats will be especially at risk of increasing water temperatures and ocean acidification brought on by climate change (Leo et al., 2018). Eggs incubated in experimental conditions set to match future projections of temperature and CO₂ demonstrated more malformations and smaller-sized larvae at hatching than those incubated in conditions more similar to the current temperature and CO₂ levels (Leo et al., 2018). Similarly, experimental evidence suggests that future levels of anticipated ocean acidification will cause organ damage, smaller size, stunted development, and poor condition of Atlantic Herring larvae, with negative impacts on larvae increasing in association with higher CO₂ levels (Frommel et al., 2014). Under poor growth conditions, smaller larvae will likely remain vulnerable to predation longer, affecting recruitment rates to older life phases (Frommel et al., 2014).

There is still high uncertainty about how Atlantic Herring will be affected by different levels of CO₂. One study found that swimming speed and foraging behavior in Atlantic Herring larvae were not affected when exposed to high levels of dissolved CO₂ (Maneja et al., 2014). However, another experiment found higher temperatures alone drove slower growth rates, while increased CO₂ had a negligible effect on size and survivorship, and food availability also played a large role (Sswat et al., 2018a). Similarly, larval swimming speed and searching behavior decreased under conditions of warmer temperatures and reduced food availability compared to similar temperatures but higher food availability (Allan et al., 2022). Community-level experiments suggest that Atlantic Herring larvae may benefit indirectly from increased CO₂, due to increases in primary production of their primary prey - plankton, which increased larvae survival by ~19% (Sswat et al., 2018b).

These studies suggest a complicated relationship between increasing ocean acidification and the survivorship of Atlantic Herring larvae, where there will likely be trade-offs between negative and positive effects.
Black Sea Bass

*Shifts in Range or Depth*

Between the periods of 1974-1977 to 2019-2022, juvenile and adult Black Sea Bass shifted their spring range across the Northeast continental shelf region by 2.24 degrees (249.72 km) north and expanded by 0.24 degrees (26.85 km). Their fall range has also shifted north by 2.04 degrees (227.33 km) but has experienced a range contraction of 1.65 degrees (183.55 km) from 1974-1976 to 2019-2022. Black Sea Bass shifted to shallower waters in spring and fall by 5.8 meters and 17.5 meters, respectively, over historical periods (NOAA Fisheries, 2022).

Distributional shifts of adult and larval life phases of Black Sea Bass were out of synchrony between 1977-1987 and 1999-2008, with larval life phases shifting northwards and inshore and adults showing no detectable change across the entire Northeast continental shelf region (Walsh et al., 2015).

Black Sea Bass are projected to experience significant gains of relative biomass during spring and fall by 2050 across the Northeast continental shelf with larger increases in the Gulf of Maine compared to the Southern New England – Mid Atlantic region under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is medium certainty that Black Sea Bass will shift the centroid of their range across the Southeast continental shelf by 65.8 km under RCP 2.6 and low uncertainty they will shift 528.6 km under RCP 8.5 by the end of the century (2081-2100), with projected changes of available thermal habitat of -2.9% under a RCP 2.6 and 82.5% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018). Black Sea Bass have been spawning further and further north over the past 40 years (McBride et al., 2018), starting to move north in Massachusetts from Buzzards Bay to Cape Cod Bay around 2000, up to Cape Ann starting around 2010, and more recently, up to the Gulf of Maine (McBride et al., 2018). Age-0 Black Sea Bass have shifted 0.021 decimal degrees north every year since 1978 --- resulting in approximately a 1° shift northward, with a strong correlation between sea temperature in the fall and maximum latitude (McBride et al., 2018). Between 2004 and 2016, the number of juvenile Black Sea Bass collected in the Long Island Sound off the coast of Milford, Connecticut, went from zero to rising steadily until the end of the study in 2016 (Mercaldo-Allen et al., 2020), with a similar increase in trawl survey data nearby; shifts corresponded with increasing average summer water temperatures off of Milford, Connecticut (Mercaldo-Allen et al., 2020). Black Sea Bass may be sensitive to rapid changes in
oceanographic conditions, as was observed at experimental sites of the Maryland coast in the southern Mid-Atlantic Bight after tropical storm Hermine in 2016; approximately half of existing Black Sea Bass in the area were observed to migrate away from the areas after the water column de-stratified (with the temperature at the bottom rising 10°C in 10 hours during the storm) and the individuals that remained demonstrated a prolonged decrease in activity (Secor et al., 2019).

Changes in Morphology or Physiology

In experimental tests, Black Sea Bass embryos appeared to be resilient to elevated ocean acidification projected under climate change scenarios; however, in some years of the experiment, highly elevated dissolved carbon dioxide levels led to increased anomalies in developing embryos (Meseck et al., 2022). In another experiment, water temperatures higher than 24°C (which are projected in part of the southern part of their range within the next 80 years) led to decreased muscle contraction performance despite acclimatization, suggesting Black Sea Bass are likely to keep shifting their range further north to find more suitable habitat conditions (Slesinger et al., 2019).

Bluefin Tuna

Shifts in Range or Depth

Bluefin Tuna were collected off the coast of Greenland for the first time in 2012 due to warming sea temperatures and the abundance of one of their main prey species (Mackenzie et al., 2014), the Atlantic Mackerel (Scomber scombrus), which has recently expanded its range north as well (Astthorsson et al., 2012).

From 1993 to 2020, warming sea surface temperature has led to Bluefin Tuna in the Atlantic increasing the total area of their range by 96 km² per year and shifting their biomass center an average of 2 km per year north and 3 km per year east (Hansell et al., 2022). Models incorporating climate projections based on the highest emissions scenario (RCP 8.5) suggest this trend will continue, with Bluefin Tuna shifting their northern range limit 6.38 decimal degrees in the western Atlantic by 2099, though their abundance is expected to decrease throughout their range (Erauskin-Extramiana et al., 2022).
**Shifts in Phenology**

Bluefin Tuna mostly spawn in the Gulf of Mexico and have a short spawning window due to rising temperatures in May and June; however, by 2090, warming seas may shift their spawning window to March and April but may severely limit the available habitat (Muhling et al., 2014). While spawning habitat will likely decrease and metabolic stress may increase in the sub-tropics, the Bluefin Tuna may be able to take advantage of a more suitable habitat further north (Muhling et al., 2016).

**Changes in Morphology or Physiology**

Bluefin Tuna larvae experimentally exposed to varying degrees of salinity and pH, conditions that may occur in the future due to climate change, had lower rates of survival in extreme highs and lows in salinity but had similar survival rates in different levels of pH (Ruiz-Jarabo et al., 2022).

**Bluefish**

**Shifts in Range or Depth**

Long-term bottom trawl survey data for Bluefish across the continental shelf of the Northeast are limited to the fall season but show a geographic range shift of 0.86 degrees (96.04 km) north and a contraction of 2.21 degrees (245.19 km). Bluefish also moved to shallower waters by 2.6 meters between 1974-1976 and 2019-2022 (NOAA Fisheries, 2022). Bluefish are projected to experience significant gains of relative biomass during spring and fall by 2050 across the Northeast continental shelf, with larger increases in the Gulf of Maine compared to the Southern New England – Mid-Atlantic regions under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Bluefish are projected to shift the centroid of their range across the Southeast continental shelf by 148.5 km under RCP 2.6 and 582.0 km under RCP 8.5 by the end of the century (2081-2100), with projected gains of available thermal habitat of 21.9% under RCP 2.6 and 358.0% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).
**Tautog**

*Shifts in Range or Depth*

Tautog are projected to experience significant gains of relative biomass during spring and fall by 2050 across the Northeast continental shelf under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). Increases in relative biomass are projected in the Gulf of Maine, while losses are projected in the Southern New England – Mid-Atlantic region.

*Shifts in Phenology*

The seasonal timing of larval Tautog occurrence shifted earlier during summer in the Mid-Atlantic Bight to Georges Bank regions between 1977-1987 and 1999-2008 (Walsh et al., 2015). Tautog expanded their summer residence time in Narragansett Bay by 37.7 days between 1959-2016; however, the change was not statistically significant and could not be definitely attributed to water temperatures, possibly due to insufficient sampling (Langan et al., 2021).

**Weakfish**

*Shifts in Range or Depth*

Range shift data for Weakfish do not exist for the Northeast region, but data from the South Atlantic region shows that juveniles and adults have shifted 1.5 degrees (167.39 km) south during spring and expanded their range by 0.74 degrees (81.84 km) between the periods 1989-1991 and 2017-2019. During fall, Weakfish shifted 2.05 degrees (228.1 km) north and contracted its range by 0.36 degrees (40.72 km). Further, during summer, Weakfish showed a 0.53 degrees (59.28 km) shift in range to the south, and its range expanded by 0.47 degrees (52.34 km).

Weakfish in the South Atlantic showed a 0.6 meters shift to shallower depths during spring, 1.6 meters deeper during fall, and 1 meter shallower during summer (NOAA Fisheries, 2022). Distributional shifts of adult and larval life phases of Weakfish were out of synchrony between the periods of 1977-1987 and 1999-2008, with adult life phases shifting northwards during fall and larval life phases showing no detectable change across the Northeast continental shelf region (Walsh et al., 2015).

Weakfish are projected to experience significant gains of relative biomass during spring and fall by 2050 across the Northeast continental shelf, with much larger increases in the Gulf of Maine compared to the Southern New England – Mid-Atlantic region under mean model
projections using the RCP 8.5 scenario (Allyn et al., 2020). There is high certainty that Weakfish is projected to shift the centroid of their range across the Southeast continental shelf by 141.0 km under RCP 2.6 and low uncertainty they will shift 826.5 km under RCP 8.5 by the end of the century (2081-2100), with projected changes of available thermal habitat of -20.3% under a RCP 2.6 and 384.0% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

**Shifts in Phenology**


**Winter Flounder**

**Shifts in Range or Depth**

Between the periods of 1974-1977 and 2019-2022, Winter Flounders shifted their spring range across the Northeast continental shelf by 0.67 degrees (74.75 km) north and contracted its range by 0.8 degrees (89.48 km). Its fall range also shifted by 0.34 degrees (38.15 km) north and contracted by 1.85 degrees (205.08 km). During spring, Winter Flounders shifted to deeper waters by 16 meters and 11.3 meters deeper during the fall (NOAA Fisheries, 2022). In the inshore waters of the Chesapeake Bay, Winter Flounders shifted their range out of the area and moved north as winter temperatures increased (Bell et al., 2018).

Winter Flounders are projected to experience significant losses of relative biomass during spring and fall by 2050 across the Northeast continental shelf with larger decreases in the Southern New England – Mid-Atlantic compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Winter Flounder is projected to shift the centroid of their range across the Northeast continental shelf by 96.3 km under RCP 2.6 and medium uncertainty they will shift 394.4 km under RCP 8.5 by the end of the century (2081-2100), with projected losses of available thermal habitat of -18.9% under a RCP 2.6 and -68.9% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).
Shifts in Phenology

The seasonal timing of the occurrence of larval Winter Flounder shifted earlier during winter in the Southern New England to Gulf of Maine regions between 1977-1987 and 1999-2008 (Walsh et al., 2015). From 1972 to 2017, peak Winter Flounder larval density in Mount Hope Bay in Rhode Island shifted approximately two weeks earlier and was strongly correlated with temperature (McManus et al., 2021).

Population Changes

Since the 1990s, Winter Flounder have experienced progressively lower levels of recruitment due to warming winter temperatures and shorter winter duration in East Coast estuaries, which has decreased the available habitat and refuge from predators for young flounders (Bell et al., 2014), though not all studies found a connection between changes in temperature and Winter Flounder decline (Frisk et al., 2018). Future population models that take into account increasing winter temperatures also confirm reduced recruitment of early life stages of Winter Flounder due in part to higher natural mortality rates (Bell et al., 2018) resulting from greater spatial and temporal overlap with predators (Howell et al., 2016; McManus et al., 2021; Langan et al., 2022). Multiple authors have raised concerns that declines in recruitment and productivity are severe enough that even prohibiting fishing of Winter Flounder may be insufficient to restore their populations to historic levels (Bell et al., 2014; Langan et al., 2022). In the southern part of their range off the coast of New Jersey, data from 1980 – 2010 indicated that while climate change played an indirect role in decreasing recruitment, a critical springtime temperature threshold affected population dynamics by increasing larval metabolism rates; higher metabolism is associated with higher and unsustainable feeding demands, especially in conjunction with increased predation pressure (Able et al., 2014). The abundance of Winter Flounder has decreased steadily in Norwalk Bay, Connecticut, in the Long Island Sound between 1990 and 2016 in association with increasing water temperatures and hypoxic conditions (Crosby et al., 2018). These results are consistent with laboratory and simulation studies showing depressed recruitment and population biomass in response to higher dissolved CO₂ combined with increasing water temperatures (Huebert et al., 2021).
**Yellowtail Flounder**

*Shifts in Range, Elevation, or Depth*

From 1974-1977 to 2019-2022, juvenile and adult Yellowtail Flounders exhibited a range shift across the Northeast continental shelf region during spring of 1.5 degrees (166.99 km) north and contracted its range by 0.59 degrees (65.93 km). Shifts in the fall range distribution were 1.12 degrees (124.4 km) north and contracted by 0.01 degrees (2.2 km). Yellowtail Flounder also shifted 25.1 meters deeper during spring and 0.5 meters during fall (NOAA Fisheries, 2022). Larval Yellowtail Flounder shifted their distribution northward between the periods of 1977-1987 and 1999-2008 in the Mid-Atlantic Bight to Georges Bank regions (Walsh et al., 2015). Over the same period, the adult and larval life phases of Yellowtail Flounder exhibited synchronous distributional shifts northward during spring across the Northeast continental shelf region (Walsh et al., 2015).

Yellowtail Flounders are projected to experience a significant loss of relative biomass during fall by 2050 across the Northeast continental shelf with larger declines in the Southern New England – Mid-Atlantic compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Yellowtail Flounder is projected to shift the centroid of their range along the coast of Eastern Canada by 209.0 km under RCP 2.6 and medium uncertainty they will shift 710.1 km under RCP 8.5 by the end of the century (2081-2100), with projected losses of available thermal habitat of -0.3% under a RCP 2.6 and -42.4% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

*Shifts in Phenology*

The seasonal timing of occurrence of larval Yellowtail Flounder shifted earlier during spring in the Mid-Atlantic Bight to Georges Bank regions between the periods of 1977-1987 and 1999-2008 (Walsh et al., 2015).

**ELASMOBRANCHS**

**Barndoor Skate**

*Shifts in Range or Depth*

Between the periods of 1974-1977 and 2019-2022, Barndoor Skates shifted and expanded their spring range across the Northeast continental shelf by 0.35 degrees (39.96 km) south and
2.24 degrees (248.49 km), respectively. Barndoor Skates also shifted and expanded their fall range by 0.04 degrees (4.94 km) north and 0.77 degrees (85.43 km) from 1974-1976 to 2019-2022. Over the same periods, Barndoor Skate moved 70.9 meters into deeper waters during spring and 2.3 meters shallower during fall (NOAA Fisheries, 2022). Barndoor Skates were projected to experience a significant loss of relative biomass during fall by 2050 across the Northeast continental shelf, with higher decreases projected in the Southern New England – Mid-Atlantic region compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). Barndoor Skates are projected to shift the centroid of their range along the coast of Eastern Canada by 279.2 km under RCP 2.6 and 875.6 km under RCP 8.5 by the end of the century (2081-2100), with projected changes in available thermal habitat of 152.8% under a RCP 2.6 and -86.8% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

Smooth Skate

Shifts in Range or Depth

Between the periods of 1974-1977 and 2019-2022, juvenile and adult Smooth Skate shifted its spring range by 0.36 degrees (40.87 km) north, and its range contracted by 0.46 degrees (51.82 km). Smooth skate also shifted its fall range by 1.14 degrees (127.23 km) north and contracted its range by 0.26 degrees (29.32 km) from 1974-1976 to 2019-2022. Smooth skate shifted to deeper waters during spring by 11 meters and 2.4 meters deeper during fall over the same periods (NOAA Fisheries, 2022). Larval Smooth Skates shifted their distribution northward between the periods of 1977-1987 and 1999-2008 in the Georges Bank to the Gulf of Maine regions (Walsh et al., 2015).

Smooth skate is projected to experience significant losses of relative biomass during spring and fall by 2050 across the Northeast continental shelf, with slightly higher decreases projected in the Southern New England – Mid-Atlantic during fall and higher decreases during spring in the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Smooth Skate is projected to shift the centroid of their range along the coast of Eastern Canada by 121.6 km under RCP 2.6 and medium uncertainty they will shift 464.6 km under RCP 8.5 by the end of the century (2081-2100), with projected
gains of available thermal habitat of 10.6% under a RCP 2.6 and 2.9% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

**Thorny Skate**

*Shifts in Range or Depth*

Between the period of 1974-1977 to 2019-2022, Thorny Skate shifted its spring range across the Northeast continental shelf by 0.13 degrees (14.85 km) north and contracted its range by 0.73 degrees (81.52 km). During fall, Thorny Skates also shifted 0.21 degrees (23.51 km) north and contracted its range by 1.2 degrees (133.62 km). Thorny Skates shifted its spring depth distribution by 8.1 meters shallower from 1974-1977 to 2019-2022 and its fall depth by 4.9 meters deeper from 1974-1976 to 2019-2022 (NOAA Fisheries, 2022).

Thorny Skate is projected to experience significant losses of relative biomass during spring and fall by 2050 across the Northeast continental shelf, with higher decreases projected in the Southern New England – Mid-Atlantic compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Thorny Skates are projected to shift the centroid of their range across the Northeast continental shelf by 118.2 km under RCP 2.6 and medium uncertainty they will shift 442.1 km under RCP 8.5 by the end of the century (2081-2100), with projected changes of available thermal habitat of 5.5% under a RCP 2.6 and -15.5% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

*Population Changes*

Depending on the type of survey data used, models project that Thorny Skates may lose between 60-80% (trawl survey data) or 37% (longline survey data) of their Gulf of Maine and Georges Bank population under the most extreme climate change scenario (Grieve et al., 2020). However, with a more moderate climate change scenario, these numbers fall to 35-45% (trawl) or 22% (longline) percent of population declines (Grieve et al., 2020). Regardless, climate change and warming seawater will likely reduce the habitat availability of the Thorny Skate in the region.
Changes in Morphology or Physiology

In an experiment of Thorny Skates caught from the Gulf of Maine, this species was sensitive to increases in water temperature of 8°C (from present-day conditions [9°C] to the high-end projections for 2100 [15°C]) with their standard metabolic rate increasing significantly (Schwieterman et al., 2019). Additionally, increasing acidification led to lower hypoxia tolerance and increased standard metabolic rate (Schwieterman et al., 2019), demonstrating how the combination of warmer and more acidic water may have a compounding effect on the physiology of Thorny Skates.

Winter Skate

Shifts in Range or Depth

Between 1974-1977 and 2019-2022, Winter Skate shifted its spring range across the Northeast continental shelf by 0.51 degrees (56.94 km) south and expanded its range by 1.34 degrees (148.34 km). During fall, Winter Skate shifted 0.16 degrees (18.64 km) north and expanded its range by 0 degrees (0.69 km). In addition, Winter Skate shifted its spring and fall depth range by 29.6 meters and 18.6 meters deeper, respectively (NOAA Fisheries, 2022). Winter skate is projected to experience significant gains of relative biomass during spring and fall by 2050 across the Northeast continental shelf, with higher decreases projected in the Southern New England – Mid-Atlantic compared to the Gulf of Maine under mean model projections using the RCP 8.5 scenario (Allyn et al., 2020). There is low certainty that Winter Skate is projected to shift the centroid of their range across the Northeast continental shelf by 118.7 km under RCP 2.6 and medium uncertainty they will shift 455.8 km under RCP 8.5 by the end of the century (2081-2100), with projected losses of available thermal habitat of -16.6% under a RCP 2.6 and -64.4% under RCP 8.5 when compared to the period of 2006-2020 (Morely et al., 2018).

Changes in Morphology or Physiology

Due to epigenetic changes within its population within the past 7000 years, the isolated population of Winter Skates in the southern section of the Gulf of St. Lawrence has reduced its size by 45%, surviving in temperatures 10°C warmer than other populations, most notably the larger population on the Scotian Shelf (Lighten et al., 2016).
**Sandbar Shark**

*Shifts in Range or Depth*

Modeling suggests that increasing water temperatures but decreasing amounts of dissolved oxygen will negatively impact habitat quality for Sandbar Shark nurseries along the bottom of the Chesapeake Bay; however, suitable habitat throughout the rest of the water column will likely increase by the end of the century (Crear et al., 2020). Changing ocean conditions and prey fields are anticipated to negatively affect juveniles if they cannot adapt their foraging behavior accordingly (Crear et al., 2020).

**Whale Shark**

*Shifts in Range or Depth*

Models of future Whale Shark habitat suitability predict that warmer sea-surface temperatures will result in a small contraction of available habitat (decrease of 2.5 - 7.4%) in the Atlantic Ocean with a slight poleward shift by 2070 (Sequeira et al., 2013). Increasing surface temperatures will lead to higher metabolic costs of diving, resulting in an increased demand for food (Makarieva et al., 2005; Sequeira et al., 2013). Given that Whale Shark prey will also shift (Beaugrand et al., 2009), range shift predictions from habitat suitability models may underestimate the degree of future range losses and shifts.

**White Shark**

*Shifts in Range or Depth*

There has been an increase in offshore sightings of White Sharks in eastern Canadian waters, coupled with an increase of sightings off of New England waters; however, it remains uncertain whether this change in occurrence is due to increasing water temperatures, better study methods, or population recovery in response to more abundant prey (Bastien et al., 2020). Population changes in White Sharks and their prey, particularly Grey Seals (*Halichoerus grypus*), could be associated with climate change, successful conservation, or a combination of both.
Summary

Marine fishes and elasmobranchs that are regularly caught in bottom trawl surveys are some of the most data-rich RSGCN species overall (for both aquatic and terrestrial) and have sufficient data to quantify observed range shifts over the last several decades and project shifts to mid and end of the century. Species important to commercial fisheries also have relatively more information than unmanaged species. However, large pelagic highly migratory species that primarily use offshore waters of the continental shelf, particularly Basking Shark, Common Thresher Shark, Dusky Shark, Longfin Mako, Night Shark, Porbeagle Shark, Sand Tiger, Shortfin Mako, Smooth Hammerhead, Whale Shark, White Marlin, and Bluefin Tuna, are severely data deficient in terms of their responses to climate change due to a lack of regular, fishery-independent monitoring programs. For these data-poor species, new technologies and molecular techniques (e.g., Environmental DNA) may be useful to understand population changes. Collaborations among states and federal agencies that operate in offshore environments are key to establishing new monitoring programs and collecting more robust datasets. Wind energy development, marine spatial planning, and impact assessments may also provide new opportunities to detect and track data-poor species in the future.

Phenological shifts were evaluated for most marine fishes, but statistically significant changes in timing were not detected (Walsh et al., 2015; Langan et al., 2021). It has been hypothesized that multiple decades of observations collected in a limited spatial area are needed to reliably detect shifts in phenology (Staudinger et al., 2019). The requirement for relatively consistent measurements at a specific location and seasonal time period is a key difference in why phenological shifts are more difficult to detect than range shifts. Some studies reported static or statistically insignificant changes in the seasonal timing of occurrence or residence time that may, in fact, be ecologically significant (Walsh et al., 2015; Langan et al., 2021) and point to expanded monitoring needs to determine if these species are adapting in place or increasing observations through improved detections could result in more robust estimates.
E) MARINE TURTLES

Table 2.7. RSGCN marine turtle species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
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<tr>
<td>Green Sea Turtle</td>
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<td>Kemp’s Ridley Sea Turtle</td>
<td>Lepidochelys kempii</td>
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<tr>
<td>Leatherback Sea Turtle</td>
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<tr>
<td>Loggerhead Sea Turtle</td>
<td>Caretta caretta</td>
<td>17</td>
</tr>
</tbody>
</table>

Figure 2.6. *Number of scientific articles for all RSGCN marine turtle species by biological response type.*

Overview

A 2017 global review of papers focused on turtles and climate change recommendations focusing on the following areas: more data on individual species responses to climate change through skewed sex ratios, improvements to species distribution models including fine-scale climate and population-specific physiological variables and physiology, and actions that increase species resilience and mitigate climate threats (Butler, 2019).
Juvenile ocean-stage sea turtles (i.e., Green Sea Turtles, Kemp’s Ridley Sea Turtles, Loggerhead Sea Turtles) are projected to increase along the North Atlantic coast and in the Northern Atlantic Ocean based on simulated data from 1993 to 2017, making sea turtle conservation a priority along the Atlantic Coast due to climate change (Putman et al., 2020). A study testing phenological shifts of four sea turtles, two that are RSGCN (Loggerhead and Green Sea Turtles), projected that phenological shifts will likely not be enough to overcome the negative impacts due to warming sands on nesting grounds and sea water temperature from climate change (Fuentes et al., 2024).

Few studies were found on the impacts of climate change on sea turtles in the northeastern U.S.; therefore, many of the resources and information presented below are out of region but may provide useful context for how sea turtles are responding to climate impacts elsewhere and guide targets for similar studies for Northeast populations.

**Green Sea Turtle**

*Shifts in Range, Elevation, or Depth*

Potential future Green Sea Turtle range shifts while foraging in the southern Atlantic Ocean are possible with rising surface temperatures and bathymetry showing whole water temperature column warming along the South American coast (Franco et al., 2020).

*Shifts in Phenology*

Green Sea Turtle nesting phenology at Cabuyal, Costa Rica, was not affected by oceanographic conditions (sea surface temperature and chlorophyll) but did influence the number of days between successful nesting events and nesting abundance (Valverde-Cantillo et al., 2019).

*Changes in Population*

While Green Sea Turtles currently do not nest in the Northeast, this information is relevant to populations of sea turtles that use Northeastern marine waters. Researchers used a combination of remote sensing and ground truthing to estimate the impacts of sea level rise on Green Sea Turtle nests in Alagadi, Northern Cyprus, from 2012 to 2016; they projected a 33.2-43.5%, 42.3-47.0%, and 57.1-59.1% flooding of nests with sea level rise 48 meters, 0.63 meters, and 1.2 meters respectively (Varela et al., 2019). Another study on sea level rise in Turkey
revealed that, for nesting locations documented between 2016 and 2018, the extreme sea level rise scenario of 1.2 m would result in 1/3 of nesting habitat loss and 18% of nest loss due to climate change (Sonmez et al., 2021). Green Sea Turtle egg viability is reduced by less than 10% after a 1 to 3-hour nest inundation of seawater and reduced by 30% after 6 hours of inundation (Pike et al., 2015), showing that not only does losing nest habitat to sea level rise negatively impact sea turtles, but does so during short-term flooding events.

A study on ocean heatwaves, by modeling the direct impacts of nesting vs. the indirect effects of food availability, demonstrated that, over the simulated lifespan of Green Sea Turtles (80 years), decreases in food availability were stronger than direct temperature effects on turtles; overall, heatwaves occurring every five years vs. 20 years lowered female egg production by 20% over their lifetime (Stubbs et al., 2020). Another by-product of extreme heatwaves was studied in Shark Bay, Australia, between 2007-2012; seagrass and Green Sea Turtle health were assessed, and researchers found post-heat wave seagrass die-offs equated to declining Green Sea turtle health (Thomas et al., 2015). A multi-species study in Australia (during 1999-2015) investigated the indirect effect of seagrass loss due to extreme climatic events; Green Sea Turtles were negatively impacted, presumably due to their foraging needs for seagrass (Nowicki et al., 2019). The Green Sea Turtles reproductive success did not decrease following the El Nino Southern Oscillation (ENSO) in the Pacific from 2015 to 2016 (Santidrian Tomillo et al., 2020). In Costa Rica, nesting success was correlated with deeper nesting depths and lower nest temperatures. Green Sea Turtles had a higher nest success rate than Olive Ridley Sea Turtles and Leatherback Sea Turtles, whose nest success declined as they reached their higher thermal tolerance limit (Santidrian Tomillo et al., 2017). Temperature-dependent sex determination studies (both field and lab) since then have found that Green Sea Turtles have an approximate 50% male-to-female ratio at temperatures between 27.1C and 30.6C and that, due to long generational times, there is a lack of genetic adaptations to rising temperatures (Tilley et al., 2019). The authors argued that temporal and spatial shifts in these species are needed to overcome future climate change impacts.
**Indirect Effects**

A Green Sea Turtle diet review showed that sea surface temperature significantly impacted the percentage of herbivory vs. omnivory, with more animal matter consumed in cooler water and higher latitudes (Esteban et al., 2020).

**Kemp’s Ridley Sea Turtle**

*Shifts in Range, Elevation, or Depth*

One study (DuBois et al., 2020) simulated the dispersal of Kemp’s Ridley Sea Turtles in the western Gulf of Mexico relative to high or low hurricane years. Higher hurricane frequency and intensity resulted in lower dispersal distances (DuBois et al., 2020). A second study simulated dispersal during juvenile years and found Kemp’s Ridleys dispersing from the Western Gulf of Mexico around Florida and past North Carolina by age 2.5 years; authors projected that Ridleys would move even farther north with climate change (Putnam et al., 2020). Once Kemp Ridley’s Sea Turtles leave the Gulf of Mexico for the North Atlantic Ocean and the coast of the eastern U.S., very few return to the Gulf of Mexico to nest (Caillouet and Gallaway, 2020).

**Leatherback Sea Turtle**

*Shifts in Range, Elevation, or Depth*

Jellyfish (gelatinous zooplankton species) distribution and hotspots of high jellyfish abundance measured in the North Atlantic show overlapping hotspots of high abundance and habitat use by feeding Leatherbacks, one within the Gulf of St. Lawrence (Nordstrom et al., 2020). Researchers hypothesized that since the waters are warming and jellyfish are shifting their range north, Leatherbacks will do the same (Nordstrom et al., 2020).

**Shifts in Phenology**

Leatherback Sea Turtles may delay nesting due to cues like high temperatures in their foraging grounds, but the phenology of nesting was not changed due to temperature fluctuations at nesting sites (Neeman et al., 2015).

**Changes in Population**

Leatherback Sea Turtle reproductive success decreased by 19% following the El Nino-Southern Oscillation (ENSO) in the Pacific during 2015-2016, showing how climate-induced
increases in interannual climate variability harm this species (Tomillo et al., 2020). A simulation from nest temperature data between 1920-2012 at Playa Grande, Costa Rica, showed that Leatherback Sea Turtles’ temperature-dependent sex determination could withstand warmer conditions. However, the authors noted there might be a temperature threshold beyond which the sex would switch; the mean has thus far increased 1.2°C above the historical mean with no discernible change in the sex ratio (Tomillo et al., 2015). In Costa Rica, nesting success was correlated with deeper nesting depths and lower nest temperatures; Leatherback Sea Turtles had the lowest nest success rate compared with Green Sea Turtles and Olive Ridley Sea Turtles (Santidrián Tomillo et al., 2017). Another study on temperature-dependent sex determination assessed the fitness and locomotion performance of Leatherback hatchlings associated with shaded (cooler) nests vs. unshaded (warmer) nests split across 48 clutches from 2013-2015 and found that shaded nests had higher hatching success and higher fitness (Rivas et al., 2019). Even though shading increases nest success, it may still harm projected population trends under climate change. Researchers (Tomillo et al., 2021) simulated four 0.5°C increments of temperature decrease under the current average temperature (30.4°C) and modeled Leatherback populations 100 years into the future with climate scenarios increasing temperatures another four 0.5°C. They found an overall population decrease because of the lower number of females in the population (Tomillo et al., 2021). Local hatchling output projected based on climate change projections were modeled in four regions within the Atlantic sites at Sandy Point National Wildlife Refuge, U.S. Virgin Islands, and Pacuare Costa Rica in the Caribbean Sea; an increase in air temperature and a decrease in precipitation were correlated with a decrease in hatchling success (Santidrián Tomillo et al., 2015). The increase of climate-induced storms and tidal disturbances on beaches formed scrapes that prevented approximately 25% of Leatherback Sea Turtles from nesting high enough on the beach to prevent flooding events during a 2013-2014 study in Pacuare Nature Reserve, Caribbean Costa Rica (Rivas et al., 2016).

**Loggerhead Sea Turtle**

*Shifts in Range, Elevation, or Depth*

In Europe, Loggerheads have started nesting further north than previously documented, with northern hatchlings experiencing lower survival success (Maffucci et al., 2016).
Shifts in Phenology

A global study on nesting phenology for Loggerhead Sea Turtles shows that temperature-dependent responses to climate change are stronger in populations nearer to the poles than in the equator (Mazaris et al., 2013). This could hold for other behavioral responses to climate change and would be important for the Northeast, the northern range limit for this species. Additional research on Loggerhead nesting phenology showed that hatching success is greater for temperate populations and likely to increase in temperate regions as temperatures increase (Pike, 2014).

Changes in Population

Climate variables such as humidity, air temperature, and accumulated precipitation influenced hatchling production in Florida, while sea surface temperature and wind speed did not (Montero et al., 2018). Similarly, in Brazil, air temperature and precipitation were the main climatic variables that led to high nest production (Montero et al., 2019). In Turkey, sea surface temperature was a stronger predictor than air temperature of nest temperature (Girondot et al., 2015). Larger swings in temperature during incubation led to lower mass in hatchling

Loggerhead Sea Turtles (Horne et al., 2014). In a Loggerhead rookery in Cabo Verde, warmer sand temperatures increased female sex ratios, smaller hatchlings, and increased predation by ghost crabs when hatchlings left the nest for the ocean (Martins et al., 2020). From Loggerhead nests sampled in Florida at Archie Carr National Wildlife Refuge during 2007-2011, researchers projected that a combination of sea level rise from climate change and land use changes from development would decrease available nesting beach area by 43% by 2050 (Reece et al., 2013). Modeling terrestrial and aquatic environments in the Mediterranean Sea showed that anthropogenic variables and sea surface temperature over 25°C decreased nesting probability (Mancino et al., 2022). Compounding effects such as these indicate the need to protect shorelines in current and projected range areas of sea turtles. A study on the Loggerhead beach nesting habitat in Greece estimated a loss of 11-71% of the habitat due to a 0.2-1.3 meter climate-induced sea level rise prediction (Katselidis et al., 2014). In addition to losing beach habitat, sea turtle nests are at risk for increased flooding. A controlled study of the six stages of incubation found that flooding during the early and late stages (near hatchling) and long-term exposure to flooding increased hatchling mortality (Limpus et al., 2020). The impacts of sea level rise on Loggerhead Sea Turtle nests in Alagadi, Northern Cyprus, from 2012-2016 was
estimated at 36.5-44.1%, 43.3-49.4%, and 62.1-67.4% flooding of nests with rising sea level 0.48m, 0.63m, and 1.2m respectively (Varela et al., 2019).

*Indirect Effects*

Other negative impacts on Loggerhead juveniles and adults from extreme climate events, such as cold stranding, have been recorded in Mexico and California; multiple factors increased the risk of strandings, such as increased disease vulnerability, predation, entanglement, and reduced swimming abilities from cold water, reducing body temperatures below optimal thermal ranges (Salinas-Zavala et al., 2020). A multi-species study in Australia investigated the indirect effect of seagrass loss due to extreme climatic events; Loggerheads were not negatively impacted, presumably due to their generalist diet (Nowicki et al., 2019). The combination of direct and indirect effects of climate change on Loggerhead Sea Turtles’ nesting behavior is studied more than other life stages.

*Summary*

Overall, more papers were found for RSGCN marine turtles than all other RSGCN reptiles, though few studies of marine turtles occurred in the Northeast, showing a regional data gap. Sea turtles (but snakes, and land turtles as well) need more research on their biological responses to climate change, particularly studies focusing primarily on temperature-dependent sex determination and modeling future climate scenarios in habitat availability were most abundant overall. All turtles will be impacted due to their long life history and generational times, preventing them from adapting to the changing climate.

**F) BUTTERFLIES & SKIPPERS**

*Table 2.8 - RSGCN butterfly and skipper species and number of scientific articles cited in this report for each species.*

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
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<tbody>
<tr>
<td>Appalachian Grizzled Skipper</td>
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<td>Scientific Name</td>
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<td>------------------------------------------</td>
<td>-------</td>
</tr>
<tr>
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<tr>
<td>Bog Elfin</td>
<td><em>Callophrys lanoraieensis</em></td>
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<tr>
<td>Chermock’s Mulberry Wing</td>
<td><em>Poanes massasoit chermocki</em></td>
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<tr>
<td>Clayton’s Copper Butterfly</td>
<td><em>Tharsalea dorcas claytoni</em></td>
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<tr>
<td>Columbine Duskywing</td>
<td><em>Erynnis lucilius</em></td>
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<tr>
<td>Crowberry Blue</td>
<td><em>Plebejus idas empetri</em></td>
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<tr>
<td>Diana Fritillary</td>
<td><em>Argynnis diana</em></td>
<td>0</td>
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<tr>
<td>Early Hairstreak</td>
<td><em>Erora laeta</em></td>
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<tr>
<td>Edwards’ Hairstreak</td>
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<td><em>Oeneis polixenes katahdin</em></td>
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<td>Mottled Duskywing</td>
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<tr>
<td>Northern Metalmark</td>
<td><em>Calephelis borealis</em></td>
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<tr>
<td>Olympia Marble</td>
<td><em>Euchloe olympia</em></td>
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<tr>
<td>Persius Duskywing</td>
<td><em>Erynnis persius persius</em></td>
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<tr>
<td>Rare Skipper</td>
<td><em>Problema bulenta</em></td>
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Table 3. Number of scientific articles for all RSGCN butterfly and skipper species by biological response type.

<table>
<thead>
<tr>
<th>Species</th>
<th>Scientific Article Count</th>
</tr>
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<td>Regal Fritillary</td>
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<tr>
<td>Two-spotted Skipper</td>
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</tr>
<tr>
<td>West Virginia White</td>
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</tr>
<tr>
<td>White Mountain Arctic</td>
<td>0</td>
</tr>
<tr>
<td>White Mountain Fritillary</td>
<td>0</td>
</tr>
</tbody>
</table>

**Figure 2.7.** Number of scientific articles for all RSGCN butterfly and skipper species by biological response type.

**Overview**

Insects, including Lepidoptera, play an integral role in ecosystems as pollinators and food sources; studying their interactions and the indirect effects of climate change on them is critical (Harvey et al., 2022). Lepidoptera are extremely sensitive to environmental changes and are threatened by herbicides, insecticides, habitat loss, pollution, and direct and indirect effects of...
climate change (Wagner, 2021). As ectotherms, climate change can be a major physiological stressor that can cause changes to misalignment with Lepidoptera host plant and emergence, shifts in range, behavior, development, and more (Hellmann et al., 2016; Patterson et al., 2019). Species with longer developmental overwintering stages, narrower diets, or restricted ranges may be more at risk of decline or extinction due to changing temperatures and precipitation than generalist species (Patterson et al., 2019). Temperature increases have led to many species emerging earlier than host plants, with spring species advancing faster than summer species. Phenological mismatch can reduce survival due to emerging before their host plant or after the host plant senesces (Patterson et al., 2019). The early emergence of adult butterflies could mean limited availability of nectar resources, late snowstorms, and freezing temperatures (Patterson et al., 2019). Asynchrony in some Lepidoptera larvae has shown increased rates of parasitism and predation (including cannibalism) due to mismatched timing of host plant emergence and the butterfly’s life cycle (Despland, 2018). Research on the effects of climate change on Lepidoptera species is still largely understudied, and little is known about the response of many of the RCGN species listed above. More research is needed at both the fine-scale and large scale, temporally and spatially.

**Karner Blue Butterfly**

*Population Changes*

Karner Blue Butterflies are particularly vulnerable to changes in their habitat and specifically the effects of climate change on their singular host plant, Wild Blue Lupine (*Lupinus perennis*) (Patterson et al., 2019; Eskildsen et al., 2015; Miller-Rushing et al., 2010). Increased temperatures can induce early larval hatching before the emergence of the Wild Blue Lupine, leading to a higher larval mortality rate in late spring or early summer (Patterson et al., 2019). Along with mismatched emergence between host plants and larvae, drought and increased summer temperatures can cause lupine to senesce early in the season, limiting the survival rate of the second generation of Karner Blues (Patterson et al., 2019).
Shifts in Range and Elevation

Monarch butterflies have a wide distribution range, with different regions showing different responses to climate change when modeled at a broad and fine-scale level (Svancara et al., 2019). Monarch butterflies may be limited by climatic conditions and partially by the range of their obligate milkweed (Asclepias spp.) host plant (Lemoine, 2015). Lemoine (2015) projected the Monarch butterfly’s distribution alongside its host plant genus Asclepias and found that Asclepias distribution accounted for ~35% of the variation in Monarch sightings. Climate change could shift the geographic range of some Asclepias species, moving the Monarch’s host plant northward further into Canada. This shift in optimal habitat could cause monarchs to expand their range north during summer breeding seasons (Lemoine, 2015). Fine-scale modeling on western populations showed a shift inland and upward elevation of California's overwinter populations as temperatures increased (Svancara et al., 2019).

Shifts in Phenology

Research from the eastern portion of the Monarch butterfly population in Cape May, New Jersey, found that there has been a shift in the monarch migration time window by 16-19 days over a 29-year period (1992 – 2020) with an average of a six-day per decade shift (Culbertson et al., 2021). They found that this shift in migration timing is correlated with higher-than-average temperatures in September and October. Similar research done from Ontario, Canada, off the Long Point Peninsula in the Great Lakes region for a 25-year period (1995 – 2020) found no significant shift in migration time; researchers hypothesized that the Great Lakes could be creating climate change refugia (Ethier and Mitchell, 2022). Increasing temperatures could lead to a mismatch in resource availability, such as nectar, in southern migratory locations (Ethier and Mitchell, 2022).

Increased temperatures in temperate regions have been shown to increase the growth rates of Monarch caterpillars, allowing them to pupate faster (Lemoine et al., 2015). However, some studies have shown that increased temperatures during rearing have not affected body mass or larval survival rate (Lemoine et al., 2015; Soule et al., 2020).
Changes to Morphology or Physiology

Research has found that Monarch caterpillars in warm locations or lower latitudes have thinner black-pigmented lines than caterpillars in colder temperatures or higher latitudes (Tseng et al., 2022). The coloration of Monarch caterpillars is linked to larval development and adult pigmentation, which is correlated with mating success and flight distance (Davis et al., 2012; Hanley et al., 2013; Davis et al., 2007; Tseng et al., 2022). The results from this research could be used to investigate monarch population dynamics and temperature-mediated shifts (Tseng et al., 2022).

Migrating Monarch butterflies have elongated wings compared to their non-migrating counterparts (Li et al., 2016; Soule et al., 2020). Their elongated forewings aid them in their ability to fly for a long period of time and over a great distance. Monarch caterpillars reared in increased temperatures showed reduced flight ability, resulting in shorter distances and duration of flight (Faldyn et al., 2018; Soule et al., 2020). These studies found that increased rearing temperatures directly impacted the aspect ratio of the forewing, making it less elongated and similar to non-migratory Monarchs. The decrease in forewing size with elevated temperatures could reduce the ability to complete the Monarch’s annual migration by limiting the duration and distance of flight.

Population Changes

Monarch butterflies migrate over long distances with multiple generations within an annual cycle. Studying the effects of climate change on the entire annual life cycle of the species and across its entire range is important for understanding the entire Monarch population and the potential threats to them. Crossley et al. (2022) found a decline in the overwintering abundance of the Monarch butterfly in California from data collected by citizen scientists through the North American Butterfly Association network across Canada and the United States. This decline in overwintering monarch population abundance is thought to potentially be caused by exposure to agricultural glyphosate and higher-than-average ambient temperatures during migration (Crossley et al., 2022). Research on the population abundance of the Monarch butterfly has shown a difference in abundance in areas with increased precipitation and limited changes in temperature (e.g., northern Ohio) to have a positive effect on abundance compared to areas with higher ambient temperature (e.g. the northern limit of their range; Zylstra et al., 2022).
Alignment with host plant emergence and resource availability due to climate change and habitat loss could also greatly impact the abundance of overwintering populations as they migrate between overwintering and breeding locations (Zystra et al., 2022). In the northern regions of the species range, Thogmartin et al. (2017) found that annual stochastic changes in population size could be related to climatic changes in the early breeding period in northern parts of the range. They also noted a negative association between population size and an increase in warm temperatures during the early season (May 1 to 20) and a positive association between warm temperatures later (May 31 to June 9) on abundance, and as minimum temperatures increased during June, they saw a negative effect on population size (Thogmartin et al., 2017). Looking at populations in Canada, Crewe et al. (2019) found that overwintering population sizes were affected by weather conditions during spring migration and recolonization.

Weather conditions during the spring and summer seasons are extremely important when it comes to determining the size of summer populations, which is a strong indicator of the size of the winter population for that year (Zylstra, 2021).

**Indirect Effects**

The concern for phenological asynchrony between milkweed species and the Monarch butterfly has little research, but Gilmour and Kharouba (2022) found that Monarch larvae may not be negatively affected if asynchrony were to occur between Monarchs and milkweed. Their research found that milkweed plants were available throughout the Monarch’s breeding season for ovipositing, and mature larger plants did not have a negative effect on larval performance (Gilmour and Kharouba, 2022). Although Monarch larvae have shown neutral success rates when feeding on native *Asclepias* species, the asynchrony of flight and the flowering window of the milkweed plants can become mismatched with increasing temperatures (Howard, 2018). This mismatch could lead to fewer milkweed plants being pollinated and the decline of their host plant populations. Increasing temperatures speed up the growth rate of milkweed species, negatively affecting the host plant's quality. One of these negative effects is increased cardenolide concentrations in the invasive *Asclepias curassavica* (Faldyn et al., 2018). The spread of *A. curassavica* or the presence of the species in extended population ranges in future climate change scenarios could result in lower success rates due to increased cardenolide concentrations. Larval success was compared to individuals who fed on native milkweed species.
under the same climate conditions. Native milkweed had a lower cardenolide concentration when reared in warmer temperatures. Mismatched hatch and host plant emergence have been shown to affect narrow-leaf milkweed in plant size, seed production, flowering time, and more (Rasmussen and Yang, 2022).

It is projected that the overwintering habitat of the Monarch butterfly in Mexico will disappear within this century, with climate change being a contributing factor (Hellmann, 2016). The decline of Monarch overwintering habitat is attributed to the stress of drought and habitat loss (Gómez-Pineda et al., 2021). Under the RCP 6.0 scenario, these crucial montane forests could increase altitude by 300 – 400 meters (Gómez-Pineda et al., 2019). Translocation of two of the most important coniferous species in these forests indicated that these species could be successful in higher altitude locations as temperatures increased and precipitation declined to establish more overwinter habitat for Monarchs. This shift in overwintering habitat would lead to a 46-77% habitat loss (Gómez-Pineda et al., 2019).

The Monarch butterfly's fall migration takes them across Texas as they fly to Mexico during hurricane season (Ries et al., 2018). Ries et al. (2018) examined the potential effects of hurricanes on Monarch butterflies along their migratory pathway and found that, like birds, Monarchs are capable of avoiding the direct effects of hurricanes but potentially would be affected by the indirect effects and the loss of available resources, like nectar, during their journey.

**Regal Fritillary**

*Shifts in Range and Elevation*

The Regal Fritillary historically has had a distribution range expanding from the western state of Colorado to the eastern coast of the United States, primarily occupying native tall-grass and mixed-grass prairies. Over the past 30 years, their range has greatly declined due to habitat loss from haying, burning, mowing, and other grassland conversions, limiting them to patchy prairie habitat in the central plains of the United States (Post Van Der Burg et al., 2022). The distribution of Regal Fritillary in the southern portion of their range could experience a population decline as average temperatures continue to rise and precipitation stays the same (Post Van Der Burg et al., 2022). Changes in temperature and lack of precipitation in the spring and early summer could lead to early larval emergence and host plant desiccation, which has
been noted in other fritillary species, like the Glanville fritillary (Tack et al., 2015; Post Van Der Burg et al., 2022). This increase in temperature in the Great Plains and Midwest could lead to a shift northward in the distribution of the regal fritillary which Swengel and Swngel (2017) has soon a range expansion northward (Post Van Burg et al., 2022).

**Summary**

Most of the research found on species responses to climate change through the literature search focuses on Monarchs, Karner Blue Butterflies, or Lepidoptera as a general group, with research lacking for specific species. More research is needed to determine Monarch migratory behavior and whether it is driven by environmental cues such as photoperiod, causing Monarchs not to follow their host plant northward (Lemoine, 2015). If monarchs follow the *Asclepias* shift northward, it could cause longer migration distances, mismatched host plant availability, and many more effects on survival (Lemoine, 2015). There is a need for time-series data to be collected for species to assess population responses to climate change on a temporal and spatial scale (Wagner, 2021). Citizen science data and museum records could be used to help assess missing information for species on the RCGN list, especially butterflies (Hellmann et al., 2016).

**G) MARINE MAMMALS**

**Table 2.9.** RSGCN marine mammal species and number of scientific articles cited in this report for each species.

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<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
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<tr>
<td>Harbor Porpoise</td>
<td><em>Phocoena phocoena</em></td>
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<td>Sperm Whale</td>
<td><em>Physeter macrocephalus</em></td>
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<td>Sei Whale</td>
<td><em>Balaenoptera borealis</em></td>
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<tr>
<td>Blue Whale</td>
<td><em>Balaenoptera musculus</em></td>
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Overview

Marine mammals, particularly the large migratory whales and porpoises that comprise the Northeast RSGCN, are experiencing high rates of ocean warming, acidification, and other climate impacts to affect their distribution, phenology, and population dynamics (Letrick et al., 2023). Relatively few studies of marine mammal responses to climate change exist in part due to the difficulties in detecting and tracking their populations over the broad spatial scales that constitute their geographic ranges. Several of the large migratory whale RSGCN have shown shifts in range and phenology over recent decades; however, the rate, magnitude, and direction of these shifts vary widely across sub-regions of their range and studies (Ramp et al., 2015; Charif et al., 2020; Pendleton et al., 2022). These conflicting patterns of earlier or later arrival may be

<table>
<thead>
<tr>
<th>Species</th>
<th>Scientific Articles</th>
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<tr>
<td>Humpback Whale</td>
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<td>North Atlantic Right Whale</td>
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</tbody>
</table>

Figure 2.8. Number of scientific articles for all RSGCN marine mammal species by biological response type.
due to the different methods used to monitor populations (e.g., acoustics vs aerial visual observations) with different detection abilities and spatial coverages.

Shifts in the distribution of marine mammals are largely attributed to the indirect effects of climate change, as prey availability, abundance, and quality are altered due to ocean warming and acidification. Planktivorous species like North Atlantic Right Whales, which depend on seasonally reliable concentrations of highly nutritious and lipid-rich Calanoid copepods, are more sensitive to bottom-up changes in primary and secondary production compared to piscivorous whales (Pendleton et al., 2022). However, rapid warming in the Gulf of Maine is having broad impacts on regional food webs with ramifications not only for whales but many other marine fish and seabird RSGCN that consume zooplankton or planktivorous forage fish such as sand lance and herring (Staudinger et al., 2019; Pershing et al., 2021). Most notably, increasing SSTs are negatively affecting cold-adapted subpolar copepod species, including *Calanus finmarchicus* through decreased growth, productivity, and shifts in overwintering phenology schedules (Staudinger et al., 2019; Staudinger et al., 2020; Pershing et al., 2021).

As food resources become increasingly unpredictable, migratory whales are occurring in unexpected locations and at unexpected times or showing elevated mortality and stranding rates, as indicated by the declaration of Unusual Mortality Events in the region (NOAA, 2024). This increases the potential for conflict and the incidence of injury and mortality of these already at-risk and endangered species with human activities such as shipping, fishing, and wind energy development.

**Fin Whale**

*Shifts in Range, Elevation, or Depth*

From 2010 to 2017, the centroid of the Fin Whale’s core range shifted 154 km in the spring, 162 km in the summer, 223 km in the fall, and 33 km in the winter, all in a northeastern direction along the U.S. Atlantic coast (Chavez-Rosales et al., 2022).

---

**Shifts in Phenology**

Over the last two decades (1998-2018), Fin whales appear to have shifted the timing of peak habitat use in Cape Cod Bay earlier by 5.8 days, although this trend is not statistically significant (Pendleton et al. 2022). This shift in timing was positively associated with the earlier onset of spring in Gulf of Maine waters. In contrast, for 1984-2010, Fin Whales arrived a total of 28 days earlier, or a little over 1 day/year earlier, in the Gulf of St. Lawrence; this shift in phenology coincided with changes in ice-free weeks and sea-surface temperatures in the region (Ramp et al., 2015). Fin Whales also left the Gulf of St. Lawrence 11 days earlier from 1984 – 2010, resulting in longer residence times in the Gulf of St. Lawrence overall (Ramp et al., 2015).

**Harbor Porpoise**

*Shifts in Range, Elevation, or Depth*

Along the northeastern U.S. coast from 2010 to 2017, the centroid of the Harbor Porpoise’s core range shifted 17 km southwest in the spring, 3 km Northeast in the summer, 10 km southwest in the fall, and 397 km Northeast in the winter (Chavez-Rosales et al., 2022).

**Humpback Whale**

*Shifts in Range, Elevation, or Depth*

From 2010 to 2017, along the Atlantic Coast of the northeastern U.S., the centroid of the Humpback Whale’s core range shifted 17 km to the south in both the spring and summer, whereas their fall range shifted 14 km to the northwest, and their winter range shifted 3.9 km to the southwest (Chavez-Rosales et al., 2022).

*Shifts in Phenology*

Humpback Whales shifted their peak habitat use in Cape Cod Bay 19.1 days earlier from 1998 to 2018, an average of 0.96 days/year (Pendleton et al., 2022). Humpbacks arrived in the Gulf of St. Lawrence 28 days earlier in 2010 than they did in 1984 and left around 27 days earlier during the same time frame, remaining in the Gulf for about the same amount of time each year (Ramp et al., 2015).
Changes in Population

Climate models exploring shifts in sea surface temperature due to climate change project that by 2100, anywhere between 35-67% (scenario RCP 4.5 and RCP 8.5, respectively) of all global Humpback Whale breeding grounds will be warmer than their current range of breeding temperatures (von Hammerstein et al., 2022).

North Atlantic Right Whale
Shifts in Range, Elevation, or Depth

From 2004 – 2014, North Atlantic Right Whales appeared less prevalent in the Gulf of Maine but more common in the Mid-Atlantic and Southern New England, though the study did not evaluate the causation of the shift (Davis et al., 2017; O’Brien et al., 2022). Models using climate change projections (specifically RCP 4.5 and 8.5) show that much of the Gulf of Maine will be unsuitable habitat for North Atlantic Right Whales in late summer and fall, except for some areas along the Scotian Shelf (Ross et al., 2021).

Shifts in Phenology

North Atlantic Right Whales appear to be leaving Massachusetts Bay to feed in Cape Cod Bay 40 days earlier each late-winter / early spring between 2008 – 2011, though results are variable (the trend reversed the last two years of the study), which may be due to warming conditions in the northwestern Atlantic Ocean (Charif et al., 2020). Moreover, another study found a similar pattern; earlier spring transition dates have driven earlier movements between habitats from 1998 - 2017, indicating that North Atlantic Right Whales may respond to temperature to start migration to Cape Cod Bay (Ganley et al., 2022). Peak habitat use of North Atlantic Right Whales in Cape Cod Bay shifted 18.1 days earlier from 1998 – 2018 and was negatively related to the earlier onset of spring in the Gulf of Maine (Pendleton et al., 2022). North Atlantic Right Whales are also staying longer in Cape Cod Bay later in the spring season, albeit with high interannual variability (Pendleton et al., 2022).

Shifts in annual and seasonal migration patterns of North Atlantic Right Whales have been linked to changes in the distribution and availability of their primary copepod prey, Calanus finmarchicus (Pershing et al. 2021). Further, recent population changes and decreased calving rates have been linked to the higher energetic costs associated with longer search times
for optional foraging grounds and lower quality food resources associated with warming waters in the region (Staudinger et al., 2020; Pershing et al., 2021; Pendelton et al., 2022),

**Sei Whale**

*Shifts in Range, Elevation, or Depth*

From 2010 to 2017, the centroid of the Sei Whale’s core range shifted 70 km in the spring, 97 km in the summer, 134 km in the fall, and 179 km in the winter, all in a southwestern direction along the northeastern U.S. Atlantic Coast (Chavez-Rosales et al., 2022).

**Sperm Whale**

*Shifts in Range, Elevation, or Depth*

From 2010 to 2017, the centroid of the Sperm Whale’s core range shifted 114 km in the spring, 202 km in the summer, 255 km in the fall, and 71 km in the winter, all in a northeastern direction along the northeastern U.S. coast (Chavez-Rosales et al., 2022).

**Summary**

Consistent and expanded distribution monitoring of whale and porpoise populations is needed to better track populations over time and space and predict occurrence, especially as wind energy development areas are pursued and a better understanding of the distribution and occurrence of key prey populations and their responses to climate change. The region's most important and least understood populations that are shared prey across whales and other RSGCN include *Calanus* copepods, *C. finmarchicus* in particular, and forage fishes, including Sand Lance, Atlantic Herring, and river herring species. Studies on the physiological responses and energetic demands associated with climate-induced shifts in distribution, phenology, and foraging behaviors in marine mammals will help fill key knowledge gaps and increase understanding of changes in population changes due to altered growth and survival rates. Model projections of climate-induced distribution and phenology changes paired with current and future fishing areas, shipping corridors, and wind energy development areas can aid marine mammal conservation efforts by identifying intersecting spatiotemporal risk areas under different closure and operation schedules scenarios.
**H) CARNIVORES & SMALL MAMMALS**

Table 2.10. RSGCN carnivore and small mammal species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>CARNIVORES</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Canada Lynx</td>
<td><em>Lynx canadensis</em></td>
<td>7</td>
</tr>
<tr>
<td>Eastern Spotted Skunk</td>
<td><em>Spilogale putorius</em></td>
<td>0</td>
</tr>
<tr>
<td><strong>SMALL MAMMALS</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tuckahoe Masked Shrew</td>
<td><em>Sorex cinereus nigriculus</em></td>
<td>0</td>
</tr>
<tr>
<td>Rock Vole</td>
<td><em>Microtus chrotorrhinus</em></td>
<td>0</td>
</tr>
<tr>
<td>Virginia Northern Flying Squirrel</td>
<td><em>Glaucymys sabrinus fuscus</em></td>
<td>6</td>
</tr>
<tr>
<td>Delmarva Fox Squirrel</td>
<td><em>Sciurus niger cinereus</em></td>
<td>0</td>
</tr>
<tr>
<td>Allegany Woodrat</td>
<td><em>Neotoma magister</em></td>
<td>1</td>
</tr>
<tr>
<td>Eastern Fox Squirrel</td>
<td><em>Sciurus niger vulpinus</em></td>
<td>0</td>
</tr>
<tr>
<td>Southern Rock Vole</td>
<td><em>Microtus chrotorrhinus carolinensis</em></td>
<td>0</td>
</tr>
<tr>
<td>Block Island Meadow Vole</td>
<td><em>Microtus pennsylvanicus provectus</em></td>
<td>0</td>
</tr>
<tr>
<td>Northern Bog Lemming</td>
<td><em>Synaptomys borealis sphagnicola</em></td>
<td>0</td>
</tr>
</tbody>
</table>
Figure 2.9. Number of scientific articles for all RSGCN carnivores and small mammal species by biological response type.

Overview

Little research has focused on how northeastern RSGCN non-volant terrestrial mammals have or may respond to climate change. Only 3 of the 11 species had any research, with most of it focused on range shifts (Figure 2.9), which show more southerly species pushing up into the ranges of RSGCN species and displacing them (e.g., Canada Lynx [Peers et al., 2013] and Northern Flying Squirrels [Wood et al., 2016]).

Canada Lynx

Shifts in Range, or Elevation, or Depth

In Ontario, Canada, Canada Lynx shifted their range over 175 km north between 1972 and 2010 (Koen et al., 2014). Due to a projected decline in snowpack in the northeastern US, increasing competition with Bobcats (*Lynx rufus*) may lead to further population declines along the southern edges of the Canada Lynx range and a more northerly and eastern shift into the future (Peers et al., 2013; Sirén et al., 2021), which is similar to model predictions of their future distribution in the Northeast (Peers et al., 2014; Sirén et al., 2022). Similar losses along the
southern edge of their range are projected in Washington state (King et al., 2020) and the northern Rocky Mountains (McKelvey and Buotte, 2018).

Indirect Effects

Due to a correlation between a gradient of snow and precipitation and the genetic structure of the Canada Lynx, dispersal limitations among subpopulations due to habitat imprinting may lead to increasingly isolated populations by 2070 (Row et al., 2014). A resulting loss in genetic diversity may further reduce the Lynx’s ability to adapt to changing conditions along the southern part of their range (Koen et al., 2014).

Virginia Northern Flying Squirrel

Shifts in Range, and Elevation, or Depth

While not specifically about the Virginia Northern Flying Squirrel, the Southern Flying Squirrel (*G. volans*) has completely displaced the nominate species (*G. sabrinus*) between 1986 and 2004 in a study area in Maine (Wood et al., 2016) as the Southern Flying Squirrel has expanded its range north. A warming climate appears to be one of many reasons for this displacement. Increased minimum summer temperatures have allowed the pathogenetic nematode (*Strongyloides robustus*) to become more infectious (Wetzel and Weigl 1994). These nematodes are more virulent in Northern Flying Squirrels than in Southern Flying Squirrels which carry them (Krachbaum et al., 2010; Wood et al., 2016). In Maine, experimental tests and field studies found that high ambient temperatures did not cause the nominate Northern Flying Squirrels heat stress (Gudde, 2022), giving further evidence that a northerly range shift is likely resulting from due to being out-competed by the Southern Flying Squirrel and introduction and higher survival rate of the nematodes it carries. Increasing temperatures at higher elevations will likely have similar effects on the Virginia Northern Flying Squirrels in the isolated, high-elevation populations that persist in Virginia and West Virginia as they are forced further up the mountains.

Indirect Effects
As Southern Flying Squirrels continue to move into new areas due to warmer winters, they have been hybridizing with the Northern Flying Squirrels in Pennsylvania (Garroway et al., 2010), which may also become a problem for the Virginia Northern Flying Squirrels.

Allegheny Woodrat

Chances in Population

Warmer spring temps appear to lead to lower female Allegheny Woodrat abundance in West Virginia, but the mechanism is unclear (Manjerovic et al., 2009).

Summary

Like many other taxonomic groups, little is known about how these mammals will likely respond to climate change and how these changes will affect the ecosystems they presently live. Further, little is even known of the basic biology of RSGCN small mammals, including their ranges and habitat requirements, so this group represents a fertile ground for further study.
I) BATS

Table 2.11. RSGCN bat species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eastern Red Bat</td>
<td>Lasiurus borealis</td>
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</tr>
<tr>
<td>Eastern Small-footed Bat</td>
<td>Myotis leibii</td>
<td>0</td>
</tr>
<tr>
<td>Hoary Bat</td>
<td>Lasiurus cinereus</td>
<td>1</td>
</tr>
<tr>
<td>Indiana Bat</td>
<td>Myotis sodalis</td>
<td>3</td>
</tr>
<tr>
<td>Little Brown Bat</td>
<td>Myotis lucifugus</td>
<td>2</td>
</tr>
<tr>
<td>Northern Long-eared Bat</td>
<td>Myotis septentrionalis</td>
<td>1</td>
</tr>
<tr>
<td>Silver-haired Bat</td>
<td>Lasionycteris noctivagans</td>
<td>2</td>
</tr>
<tr>
<td>Tricolored Bat</td>
<td>Perimyotis subflavus</td>
<td>2</td>
</tr>
<tr>
<td>Virginia Big-eared Bat</td>
<td>Corynorhinus townsendii virginianus</td>
<td>0</td>
</tr>
</tbody>
</table>

Overview

Little information was found about the biological responses of the Northeast’s bat RSGCN to climate change. However, bats in the northeastern U.S. will likely be especially sensitive to climate change because much of their biology, including reproduction and hibernation, is closely related to temperature (Loeb and Winters, 2013). For instance, increasing winter temperatures will likely cause warmer microsites where bats hibernate (Boyles et al., 2017), and variation in ambient microsite temperature increases energy expenditure by hibernating bats (Boyles and McKechnie, 2010). Hibernacula near cave entrances are quickly affected by outside temperatures, as opposed to sites deeper in more shielded caves (Boyles et al., 2017). However, even small temperature variations can significantly increase energy
expenditure across winter (Boyles and McKechnie, 2010), so areas with a large range of microclimates may hold on to their bat populations while those with a more limited range may not (Boyles et al., 2017). Despite that, some evidence suggests that ambient temperature may not be the most important factor in energy conservation during hibernation, as behavior, humidity, and initial fat mass were also important factors for Tricolored Bats (McGuire et al., 2021) and Tricolored Bats experimentally held in varying temperatures during hibernation showed no difference in metabolic rate when hibernation ended (Boyles et al., 2022).

![Figure 2.10. Number of scientific articles for all RSGCN bat species by biological response type.](image)

Variations in precipitation patterns that lead to increased drought and more extreme storms will likely affect foraging conditions for bats. Insect populations often decline during drought (Hawkins and Holyoak, 1998), resulting in increased foraging costs and decreased annual survival for bats (Loeb and Winters, 2013). Most insectivorous bats must drink to maintain water balance, and water needs increase considerably during pregnancy and lactation (Kurta et al., 1989; Adams and Hayes, 2008). As such, severe droughts, especially when combined with unusually cold or hot temperatures, will likely lead to lower reproductive success rates (Bourne and Hamilton-Smith, 2007; Adams, 2010).
Changes in precipitation patterns are already contributing to more severe wildfires across North America, specifically in boreal forests (Stephens et al., 2014). The increased risk that wildfires may impose on bat populations is not well known (Jung, 2020); though, studies evaluating the effect of smaller, prescribed burns have shown burned areas benefit bats in creating better roosting and foraging opportunities (e.g., Ford et al., 2016; O’Keefe and Loeb, 2017). While smaller prescribed burns may increase bat activity in some areas, larger wildfires can negatively impact some species (e.g., Snider et al., 2013; but see Buchalski et al., 2013). For example, after a “megafire” in the boreal forest of Yukon, Canada, despite creating more open areas for Little Brown Bats to forage, bat activity in the burned areas was much lower than in nearby unburned areas, with almost no activity occurring in burned upland areas (Jung, 2020). Importantly, sites of “megafires” in eastern Canada in the summer of 2023 may provide a good opportunity to work with Canadian colleagues to explore post-fire habitat usage.

There is still much to learn about how these ecologically important species that provide valuable ecosystem services as consumers of nuisance insects are and will be impacted by climate change. Much of the research on these bat species has focused on roosting and hibernating site selection and how these sites may influence energy budgets. However, since bats are highly mobile, one particularly large data gap exists in considering how these species will shift their geographic ranges, especially since the one species with a projected range shift (Indiana Bats) seems like it could abandon most of its summer range in the midwestern U.S. for the northeastern U.S. in the next 50 years (Loeb and Winters, 2013). Having reasonable estimates of where these species may be in the near- and long-term is crucial in developing management plans for the future. While none of the northeastern bats of greatest conservation need had much research on biological responses to climate change, we found no research at all for one-third of these species. Other major data gaps include a complete lack of published data on how climate change will affect population trends or changes in the physiology and morphology of RSGCN bats in the Northeast and how any indirect effects of climate change will impact them.
**Hoary Bats**

*Shifts in Range, or Elevation, or Depth*

Experimental evidence suggests that bat species that roost in warmer microhabitats, such as Hoary Bats in foliage, have a higher heat tolerance than those that roost in more protected areas, like Little Brown and Silver–haired Bats that roost in tree crevices (Noakes et al., 2021). Hoary Bats are known to only roost in Eastern Hemlock (*Tsuga canadensis*; Veilleux et al., 2009), which is experiencing drastic declines in the Northeast due to the Hemlock Wooly Adelgid (*Adelges tsugae*), which will expand farther in the Northeast due to climate change (Paradis et al., 2008; Ellison et al., 2018). This Wooly Adelgid expansion and subsequent die-off of Eastern Hemlock may effectively destroy much of the Hoary Bat’s roosting habitat.

**Indiana Bat**

*Shifts in Range, and Elevation, or Depth*

As mentioned above, Indiana Bats may shift their summer maternity distribution from the midwestern U.S. to the northeastern U.S. and Appalachian Mountains, with the western part of their maternity range (Missouri to Ohio) likely becoming unsuitable due to increasing temperatures during the maternity season under most future climate projections (Loeb and Winters, 2013). To reduce some of the effects of a warming climate, Indiana Bats may use local site selection (e.g., select trees under dense canopy) to mediate the effects of warming temperatures before abandoning an area (Loeb and Winters, 2013). For instance, female Indiana Bats in Missouri selected roosts deeper in the shade on days with higher temperatures in the morning and roosts more exposed to the sun on days with lower temperatures (Callahan et al., 1997), which is a similar shift in the American Pika’s (*Ochotona princeps*) thermal habitat use and behavioral adaptive capacity (Beever et al., 2016).

Managing Indiana Bat habitat in the northeastern United States and the Appalachian Mountains of the Southeast is important since these areas will likely serve as climate refugia for this species (Loeb and Winters, 2013; for more on climate refugia, see Morelli et al., 2016).

*Shifts in Phenology*

Indiana Bats have not appeared to change their arrival and departure dates to and from their summer colonies in Indiana between 1998 and 2014 very much (Petitt and O’Keefe, 2017),
which suggests that photoperiod and/or developmental physiology may be the most important
cues driving migration phenology, but more research is needed (Petitt and O’Keefe, 2017).

**Little Brown Bats**

*Shifts in Range, or Elevation, or Depth*

Though outside of the northeastern region, Little Brown Bats did not appear to alter their
elevational range in the Rocky Mountains from 1996 to 2016, despite an increase of 1.8°C over
that time period in the study area (Adams, 2017). This suggests that their flexibility in shifting
roost sites, as opposed to other myotis species in the area may be allowing them to remain at the
same elevations while others are starting to shift elevations (Adams 2017). However, less
precipitation during the summer months may lead to higher mortality rates (Frick et al., 2010)
due to decreased abundance of insects for feeding.

**Long-eared Bats**

*Shifts in Range, or Elevation, or Depth*

Much like Indiana Bats (see above), Long-eared Bats also select more shady roost sites
on warmer days and more exposed roosts on cooler days, depending on which part of their range
they are in - females switched roosting spots less when it rained in Nova Scotia than in
Kentucky but females in both places did change roosts depending on the ambient temperature
and weather conditions (Patriquin et al., 2016).

**Silver-haired Bats**

*Shifts in Phenology*

Across a 3-year study, female Silver-haired Bats arrived earlier to roost sites in warmer
years (Jonasson and Guglielmo, 2016).

**Summary**

To address the multiple data gaps with this taxa, future research could focus on how these
volant species will shift their range due to climate change. Identifying where these species may
end up in 50 years will be key in protecting the appropriate habitat as bats shift their ranges.
Given the recent increase in wildfires and highly variable drought conditions in the region,
additional research on how bats use recently burned landscapes will be valuable. We did not identify any research that explored many of the biological responses to climate change in these taxa, including changes in morphology or physiology, population shifts, or any indirect effects for any of these species. In addition to general research subjects, another large data gap exists in understanding how many bat RSGCN have been and will respond to climate change. Peer-reviewed journal articles since 2013 were only found for six of the nine species, and only Indiana Bats had substantial research exploring biological responses to climate change.

Additionally, because Indiana Bats vary in their microhabitat use throughout the summer maternity season depending on daily temperature fluctuations (Callahan et al., 1997) and varying use of heterothermy and torpor at different stages of reproduction (Bergeson et al., 2021), managers in their current range may help these bats by protecting and restoring habitats that provide maternity colonies a wide range of microhabitats (Bergeson et al., 2021). Some short-term management measures, such as modulating temperatures at specific microsites, may help maintain Indiana Bat populations (Boyles et al., 2017).

**J) FRESHWATER TURTLES**

**Table 2.12.** RSGCN freshwater turtle species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blanding's Turtle</td>
<td><em>Emydoidea blandingii</em></td>
<td>4</td>
</tr>
<tr>
<td>Bog Turtle (Northern pop.)</td>
<td><em>Glyptemys muhlenbergii</em></td>
<td>0</td>
</tr>
<tr>
<td>Diamond-backed Terrapin</td>
<td><em>Malaclemys terrapin</em></td>
<td>3</td>
</tr>
<tr>
<td>Eastern Box Turtle</td>
<td><em>Terrapene carolina</em></td>
<td>0</td>
</tr>
<tr>
<td>Northern Red-bellied Cooter (Massachusetts pop.)</td>
<td><em>Pseudemys rubriventris</em></td>
<td>0</td>
</tr>
<tr>
<td>Spotted Turtle</td>
<td><em>Clemmys guttata</em></td>
<td>1</td>
</tr>
<tr>
<td>Wood Turtle</td>
<td><em>Glyptemys insculpta</em></td>
<td>1</td>
</tr>
</tbody>
</table>
Overview

Despite being an active group for research focus in the Northeast, little is known about how RSGCN freshwater turtle species will respond to climate change, as there are only a handful of studies that have explored the subject, and only half of the species have any research at all (Figure 2.11). Further, most data on the most studied species, the Blanding’s Turtle, is outside of the Northeast region (Hamilton et al., 2018; Bryer et al., 2020). Given the importance of water features to the biology of these turtles, such as lakes and vernal pools, research and monitoring of associated aquatic habitats and studies of changing phenology, range shifts, and climate refugia may be especially important going forward to identify potential adaptation strategies for these species.

Figure 2.11. Number of scientific articles for all RSGCN freshwater turtle species by biological response type.
Blanding's Turtle

*Shifts in Range, Elevation, or Depth*

Climate change is projected to impact Blanding’s Turtle habitat more than land-use policy change across Wisconsin by 2050; in both low- and high-emission models, the habitat will shift to the northern edge of the state, where there are no current records of occurrence (Hamilton et al., 2018).

*Shifts in Phenology*

Blanding’s Turtles shifted their initiation of nesting 2.1 days earlier due to March temperatures in Wisconsin from 2002-2019 (Byer et al., 2020).

*Changes to Morphology or Physiology*

Modeled climate change variables did not impact genetic diversity or sex ratio in Wisconsin for Blanding’s Turtles, Painted Turtles (*Chrysemys picta*), or Snapping Turtles (*Chelydra serpentina*, Reid and Peery 2014). In Nova Scotia, Blanding’s Turtle growth based on plastron growth rings measured 1975-2007 showed positive correlations with temperature but not precipitation, where warmer May in the previous year and warmer August in the current year show increased growth and cooler October in the previous year show decreased growth (Richard et al., 2014).

Diamond-backed Terrapin

*Shifts in Range, Elevation, or Depth*

Sea level rise projection impacts on Diamond-backed Terrapin may cause a 99.4% loss of current habitats and a decrease of 59.1% nesting area in New Jersey by 2100 (Ganter and Christman, 2023). In Maryland, Chesapeake Bay and other coastal areas are projected to lose 25-55% of Diamond-backed Terrapin habitat in the next 10 years and 80% by 2100 due to sea level rise (Woodland et al., 2017).
Changes in Population

Rising temperatures may impact Diamond-backed Terrapin nest success; one study found that, at 34°C, hatching success decreased to 50% or less, with complete hatching failure at 37°C (Rowe et al., 2020).

Spotted Turtle

Changes in Population

From 2018-2020, Spotted Turtles were captured from Florida to Maine across 12 states, and recent climate change has likely led to lower proportions of males in sites that had them but only in areas that were on the warmer part of their range, indicating a climate-induced temperature-dependent sex ratio shift (Roberts et al., 2023).

Wood Turtle

Shifts in Range, Elevation, or Depth

Ecological niche models show that in the United States portion of the Wood Turtles range, higher latitudes in the states of Maine, New Hampshire, Vermont, and New York will hold the majority of climate refugia, while suitable habitat will decrease across the Northeast by 29-52% by the year 2070 (Mothes et al., 2020).

Summary

Turtles need more research on their biological responses to climate change, particularly studies focusing primarily on temperature-dependent sex determination and modeling future climate scenarios in habitat availability, which were most abundant overall. All turtles will be impacted due to their long life history and generational times, preventing them from adapting to the changing climate. Additional studies are needed to reduce and avoid disturbance during hibernation/brumation periods under projected warming scenarios.
K) SALAMANDERS

Table 2.13. RSGCN salamander species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
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</thead>
<tbody>
<tr>
<td>Big Levels Salamander</td>
<td>Plethodon sherando</td>
<td>0</td>
</tr>
<tr>
<td>Black Mountain Salamander</td>
<td>Desmognathus welteri</td>
<td>0</td>
</tr>
<tr>
<td>Blacksburg Salamander</td>
<td>Plethodon jacksoni</td>
<td>0</td>
</tr>
<tr>
<td>Blue-spotted Salamander</td>
<td>Ambystoma laterale</td>
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<tr>
<td>Cheat Mountain Salamander</td>
<td>Plethodon nettingi</td>
<td>0</td>
</tr>
<tr>
<td>Cow Knob Salamander</td>
<td>Plethodon punctatus</td>
<td>0</td>
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<tr>
<td>Dixie Cavern Salamander</td>
<td>Plethodon dixi</td>
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<tr>
<td>Eastern Tiger Salamander</td>
<td>Ambystoma tigrinum</td>
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<tr>
<td>Flat-headed Salamander</td>
<td>Desmognathus planiceps</td>
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<tr>
<td>Green Salamander</td>
<td>Aneides aeneus</td>
<td>2</td>
</tr>
<tr>
<td>Hellbender</td>
<td>Cryptobranchus alleganiensis</td>
<td>1</td>
</tr>
<tr>
<td>Jefferson/Blue-spotted Salamander Complex</td>
<td>Ambystoma jeffersonianum/laterale complex</td>
<td>0</td>
</tr>
<tr>
<td>Peaks of Otter Salamander</td>
<td>Plethodon hubrichti</td>
<td>0</td>
</tr>
<tr>
<td>Shenandoah Mountain Salamander</td>
<td>Plethodon virginia</td>
<td>0</td>
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<tr>
<td>Shenandoah Salamander</td>
<td>Plethodon shenandoah</td>
<td>0</td>
</tr>
<tr>
<td>Valley and Ridge Salamander</td>
<td>Plethodon hoffmani</td>
<td>0</td>
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<tr>
<td>Weller’s Salamander</td>
<td>Plethodon welleri</td>
<td>0</td>
</tr>
<tr>
<td>West Virginia Spring Salamander</td>
<td>Gyrinophilus subterraneus</td>
<td>0</td>
</tr>
<tr>
<td>Yellow-spotted Woodland Salamander</td>
<td>Plethodon pauleyi</td>
<td>0</td>
</tr>
</tbody>
</table>

Overview

Overall, climate change literature is sparse for RSGCN amphibians, particularly salamanders in the northeastern U.S, with only two types of biological responses documented in the literature search (Figure 2.12). The species richness of amphibian communities is particularly at risk to local decreases in water availability during breeding seasons and warmer winters (Miller et al., 2018). For many amphibian species, like the Green Salamander, natural history research is still needed to understand their biological responses. For example, only one study in
Tennessee was found citing climate change threats such as habitat shifts and alteration, droughts, and temperature extremes but no underlying drivers or mechanisms resolved (Niemiller et al., 2022). Global research has shown that altered precipitation regimes due to climate change can effect life history traits; for example, frog body size has historically adapted to changing temperature and precipitation rates (Sheridan et al., 2022).

**Figure 2.12.** *Number of scientific articles for all RSGCN salamander species by biological response type.*

**Blue-spotted Salamander**

*Shifts in Range, Elevation, or Depth*

Climate suitability for Blue-spotted Salamanders projected for 2050 and 2070 in the eastern United States and Canada shows that most (91%) of the current (1960-1990) range will still be suitable. Furthermore, range expansion northward could occur as suitable habitats may be available as far north as 55° N, such as Alaska and parts of the Hudson Bay region in Canada (Widmer et al., 2022).
Eastern Tiger Salamander
*Shifts in Range, Elevation, or Depth*

Shifting hydrology and precipitation influenced the wetland use of Eastern Tiger Salamanders in a multi-year (2012-2014) study in Iowa (Lannoo and Stiles, 2022). Vernal pool use shifted between wet and dry years; both Eastern Tiger Salamander and Northern Leopard Frog (*Lithobates pipiens*) occupancy decreased in seasonal and semi-permanent wetlands in dry years while occupancy increased in permanent wetlands in dry years; the opposite was true in wet years (Lannoo and Stiles, 2022). Research out of the NE CASC shows that vernal pool seasonal duration is likely to change with climate change, but there is potential for vernal pool refugia to persist long enough for amphibians to complete their breeding cycles (Cartwright et al., 2022).

Green Salamander
*Shifts in Phenology*

Shifts in seasonal temperatures disrupt Green Salamander breeding and over-wintering behaviors (Soto et al., 2021).

Hellbender
*Shifts in Range, Elevation, or Depth*

Habitat suitability models show that climate suitability will likely increase for the Eastern Hellbender in areas within West Virginia, Virginia, and Pennsylvania by 2050 and 2070 (Sutton et al., 2023).

Summary

More research and monitoring are needed to tackle climate change’s impacts on amphibians especially in critical habitats such as vernal pools and riparian zones. Studies employing integrated modeling approaches can be useful to identify optimal habitat and climate refugia areas for salamanders and other amphibians (Newman et al., 2022). In addition, Conservation Actions Plans (CAP) from the Partners in Amphibian and Reptile Conservation (PARC) have identified additional research gaps, as well as conservation implementation and inventory and monitoring strategies (Soto et al., 2021).
L) BEES

Table 2.14. RSGCN bee species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
</thead>
<tbody>
<tr>
<td>Parnassia Mining Bee</td>
<td><em>Andrena parnassiae</em></td>
<td>0</td>
</tr>
<tr>
<td>Rusty-patched Bumble Bee</td>
<td><em>Bombus affinis</em></td>
<td>0</td>
</tr>
<tr>
<td>Ashton Cuckoo Bumble Bee</td>
<td><em>Bombus ashtonii</em></td>
<td>0</td>
</tr>
<tr>
<td>Yellow-banded Bumble Bee</td>
<td><em>Bombus terricola</em></td>
<td>3</td>
</tr>
<tr>
<td>Macropis Cuckoo Bee</td>
<td><em>Epeoloides pilosulus</em></td>
<td>0</td>
</tr>
<tr>
<td>Fringed Loosestrife Oil-collecting Bee</td>
<td><em>Macropis ciliata</em></td>
<td>0</td>
</tr>
<tr>
<td>Patellar Oil-collecting Bee</td>
<td><em>Macropis patellata</em></td>
<td>0</td>
</tr>
<tr>
<td>A Mining Bee, PA</td>
<td><em>Protandrena abdominalis</em></td>
<td>0</td>
</tr>
</tbody>
</table>

Overview

Although climate-specific information on Northeast RSGCN bee species is currently limited, research on more common bee species’ responses to climate change has increased over the last decade, with most studies focused on the *Bombus* genus (Buckner and Danforth, 2022). General research on *Bombus spp.* indicates that bees are particularly vulnerable to increased temperatures, causing range shifts north in latitude and to higher elevations (Dibble et al., 2017; Feuerborn et al., 2023). Along with range shifts, studies have shown the effects of increased temperature and precipitation patterns on foraging behavior and activity, which could impact reproduction, dispersal, and survival (Drummond et al., 2017). Collecting species-specific data...
across an entire range is difficult for many reasons, including time and cost restraints in response, researchers have been increasing the use of publicly available occurrence data to help answer questions. Buckner and Danforth (2022) used publicly available occurrence data to look at the response of *Macropis nuda*, an oil-collecting bee, and its host plant, *Lysimachia ciliata*, to four climate change scenarios. They found a shift in the species range to refugial habitat along with their host plant. *M. nuda* is closely related to two species on the RSGCN list above; this information could help inform researchers on potential future research for other *Macropis* species but does not provide us with the information needed for informed decision-making.

**Figure 2.13. Number of scientific articles for all RSGCN bee species by biological response type.**

**Yellow-banded Bumblebee**

*Shifts in Range and Elevation*

There were larger percentages of Yellow-banded Bumblebee at high elevations in refugial habitats compared to lower elevation locations in the White Mountain National Forest in New Hampshire (Tucker and Rehan, 2017). These results correspond with other research on *Bombus* spp. showing range shifts (Dibble et al., 2017; Feuerborn et al., 2023).
Indirect Effects

Shifts to higher elevations to avoid increasing summer temperatures could isolate populations and restrict gene flow across the species’ distribution range (Tucker and Rehan, 2017).

Summary

Species-specific information on RSGCN bees is needed to better understand life history, ecological roles, habitat requirements, and distribution at a fine-scale level to make informed decisions to protect RSGCN bees in the Northeast region (Jacobson et al., 2018; Liczner and Colla, 2020).

M) SNAKES

Table 2.15. RSGCN snake species and number of scientific articles cited in this report for each species.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th># Research Articles</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eastern Massasauga</td>
<td><em>Sistrurus catenatus</em></td>
<td>0</td>
</tr>
<tr>
<td>Mountain Earthsnake</td>
<td><em>Virginia valeriae pulchra</em></td>
<td>0</td>
</tr>
<tr>
<td>Northern Pinesnake</td>
<td><em>Pituophis melanoleucus melanoleucus</em></td>
<td>0</td>
</tr>
<tr>
<td>Short-headed Gartersnake</td>
<td><em>Thamnophis brachystoma</em></td>
<td>0</td>
</tr>
<tr>
<td>Timber Rattlesnake</td>
<td><em>Crotalus horridus</em></td>
<td>2</td>
</tr>
</tbody>
</table>

Overview

Only two research articles covered the climate change response of five northeastern RSGCN snake species (Figure 2.14), with only one of those articles being published in the last 20 years (Panella et al., 2020). Much remains to be known about how these species may respond to climate change.
Figure 2.14. *Number of scientific articles for all RSGCN snake species by biological response type.*

**Timber Rattlesnake**

*Changes in Population*

In West Virginia, a 12-year study (1989-2001) found that reproductive failure occurs 25-35% of the time. Failure is correlated with lower temperatures and above-average cloud cover during the active months of May through September (Martin, 2002).

*Indirect Effects*

A Conservation Species Assessment in Nebraska noted that the Timber Rattlesnake is highly vulnerable to climate change, needing rocky outcrops surrounded by specific habitat characteristics that make adaptation unlikely (Panella et al., 2022).
Summary

Snakes are very data-poor and more research on their biological responses to climate change is needed. Additional studies would be particularly useful to reduce and avoid disturbance during hibernation/brumation periods under projected warming scenarios.

N) DATA POOR TAXONOMIC GROUPS

Our literature search yielded no journal articles on the biological responses to climate change for the amphibian group frogs and toads, the small mammal group rabbits and hares, terrestrial snails, multiple insect groups including fireflies, stoneflies, mayflies, and caddisflies, moths, tiger beetles, dragonflies and damselflies, and freshwater species including crayfish, bivalves, fairy, clam, and tadpole shrimp, and freshwater fishes (see Appendix 2.1 for full species list). Given that we used climate change-specific keywords (see Methods), it is possible that some related research with climate-change implications exists (e.g., thermal physiological tolerances) but was not captured. In these cases, taxonomic experts from each state engaged in the SWAP process may be able to help identify relevant research and help fill specific data gaps for these species. Given the relative lack of climate change-specific research, any effort states may undergo to learn how these species respond to climate change through in situ monitoring or studies is a high priority. For instance, understanding where populations from each of these taxa groups are declining will help shed light on whether their range is shifting and what conditions these species may be sensitive to and will help inform management actions such as protecting and maintaining specific climate change refugia or if they may need more aggressive management, such as assisted migration, to stay within their thermal tolerance range.

Management and adaptation actions that benefit other RSGCN with similar habitat needs as data-poor species can serve as value-added actions to protect or enhance resources for species with high uncertainties until more information is gathered.

One species of crayfish, the Digger Crayfish, yielded one paper noting that the species has expanded its range northwesterly in Ontario by an estimated 7620 km² between 1996 and 2020; however, the study did not test climate’s role in the range expansion (Guiaşu, 2021). Notably, this species was only recently found for the first time in Pennsylvania in 2018. The person who found it considers it native to the state (Loughman et al., 2018), so this may not
necessarily mark a range expansion. A key action here would be to verify the historical distribution in the Northeast region using museum specimens and other archival methods. One group, moths, did yield some tangential results, though not directly related to climate; most research focuses on their roles as a pest species and their effects on agriculture or defoliation in forest systems. Many of these journal articles focus on the range expansion of non-native or invasive species rather than at-risk or native moth species. Moths in their adult and caterpillar form comprise a substantial percentage of insect biomass available for insectivores such as bats and songbirds (Wagner et al., 2021). They also play an important role as pollinators and have a long history as a focal species for collectors, allowing researchers to study the long-term global effects of climate change on them (Wagner et al., 2021). Communicating their importance and need for information to researchers and the agricultural sector could help narrow the gap between research being done on common species, species related to agricultural benefits, and at-risk species (Liczner and Colla, 2020).

IV. CONCLUSIONS

Overall, our comprehensive literature search yielded no relevant research for many taxonomic groups, especially most invertebrate groups (e.g., fireflies, dragonflies, and mayflies), very little for some vertebrate groups (e.g., salamanders and small mammals), but a great deal for others (e.g., marine fish and mammals). Perhaps unsurprisingly, a large proportion of research focused on commercially harvested species, including the Atlantic Cod, and American Lobster, and “charismatic” species, including the Monarch and Green Turtles. One species, the Atlantic Salmon, was the focus of more research articles in the literature search (N=1,180) than all mammal and insect species combined (N=1,172). However, after undergoing our screening process for climate and regionally specific information, the number of papers included in this chapter was reduced to just 20 studies. Additionally, marine invertebrate species were the subject of many experimental studies, which yielded a better understanding of their physiological responses to higher water temperatures and increasing ocean acidification. A new online tool built on long-term monitoring of the continental shelf ecosystem by NOAA (DisMAP) also led to more consistently modeled information for marine species range and distributional shifts than other RSGCN groups. In contrast, few, if any, studies explored similar physiological responses and limits in terrestrial species.
Discrepancies in taxonomic and biological response research across RSGCN taxonomic groups highlight a significant need for new studies and in situ monitoring going forward in both terrestrial and aquatic systems. In Chapter 4, we present examples of how new actions can be developed or actions from the 2015 SWAPs can be enhanced by outlining different approaches and ways of thinking about prioritizing the next steps for data-rich and data-poor species. For data-rich species, we outline a step-by-step process that encourages reflection on five standard questions, known as the 5Ws approach, to integrate climate information and adaptation tools into species and habitat-specific actions through two examples using hellbender salamander and vernal pools (Oaks et al., 2022). For data-poor species, we provide demonstrations and examples of approaches to address RSGCN with little to no data or a key piece of information missing through actions that prioritize research and monitoring and threat-based vignette narratives. Additional information on data-poor RSGCN responses to climate change that did not appear in the peer-reviewed literature may be found in Climate Change Vulnerability Assessment results presented in Chapter 3 that were received through expert-elicited input to these initial adaptation planning tools.

V. LITERATURE CITED


Migratory shorebird adheres to Bergmann’s Rule by responding to environmental conditions through the annual lifecycle. *Ecography*, 42, 1–12.


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performance and increase disease susceptibility in subadult, female American lobsters Homarus americanus H. *Journal of Crustacean Biology, 40*(5), 634–646.


Lemoine, N. P. (2015). Climate change may alter breeding ground distributions of eastern migratory monarchs (Danaus Plexippus) via range expansion of Asclepias host plants. *PLOS ONE, 10*(2).


temperature resiliency in the lobster cardiac nervous system. *Frontiers in Neuroscience, 17*(1113843).


importance of coastal storms and undeveloped landscapes for the creation and maintenance of early successional habitat. *PLoS ONE*, 14(7).


CHAPTER 3: NORTHEAST REGIONAL SPECIES AND HABITATS AT GREATEST RISK AND MOST VULNERABLE TO CLIMATE CHANGE

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Key findings

- Climate change vulnerability is characterized through three components: exposure, sensitivity and adaptive capacity.
- Climate Change Vulnerability Assessments (CCVAs) are goal-oriented decision-support tools that help managers determine which species are vulnerable, how vulnerable they are, why they are vulnerable, where they are vulnerable, and when they are vulnerable.
- CCVAs help set management and planning priorities, identify primary drivers of climate vulnerability, and enable more efficient allocation of resources; however, they are only a starting point and one piece of the puzzle of developing climate-informed management actions.
- CCVAs targeting ecological systems can be focused at the species, habitat, or ecosystem level; there are different interpretations, treatments, and approaches to assessing climate vulnerability. Therefore, it is important to examine the specific factors, the definitions of vulnerability and the limitations of each study.
- Non-climate-focused risk assessments can be used in conjunction with CCVAs to provide cumulative information about species’ and habitats’ vulnerability to multiple threats and point to comprehensive actions to increase resilience and protection.
- Recent advancements in the field of CCVAs include an increase in assessments of coastal and marine systems, global-scale species assessments, more nuanced understanding of species’ adaptive capacity, and an expanded suite of factors including non-climate threats and socioecological approaches into evaluations.
- Between 2015-2023, 13 CCVAs with 315 unique assessments for 156 NE Regional Species of Greatest Conservation Need (RSGCN) and 11 CCVAs targeting associated habitats were published, including 92 species not previously captured in regionally relevant climate change vulnerability assessments. NatureServe’s Climate Change Vulnerability Index (CCVI) was the most used CCVA framework during this time period. Four new studies used the CCVI to assess fish and wildlife species across the Northeast. Other common frameworks included the Ecosystem Vulnerability Assessment Approach (EVAA) and the Sea Level Affecting Marshes Model (SLAMM), which were each used three times in habitat assessments of forests and wetlands, respectively.
Birds and Fish were the most frequently assessed taxonomic groups of all RSGCN. Crustaceans and Marine Invertebrates were the least assessed taxonomic groups and represent a gap in our current understanding of the Northeast’s RSGCN.

- Forests and Woodlands (N=10 studies) and Tidal Wetlands and Flats (N=9 studies) were the most assessed habitat classes overall. Subterranean Areas and Great Lakes were the only natural habitat classes with no CCVA results, representing a gap in understanding of their climate vulnerability in the Northeast.

The objectives of this chapter are to describe climate change vulnerability, its components, and advancements in understanding since 2015. Additionally, this chapter provides comprehensive information and considerations on Climate Change Vulnerability Assessments (CCVAs) and synthesizes different frameworks and spatial extents (e.g., states), with an explicit focus on Northeast Regional Species of Greatest Conservation Need (RSGCN) and their dependent habitats. Appendices detail individual species and natural habitat vulnerability rankings, including contextual study-specific information. Information is cumulative to include CCVA results summarized in Staudinger et al. (2015) and published as of July 2023.

I. VULNERABILITY TO CLIMATE CHANGE

A) INTRODUCTION TO CLIMATE VULNERABILITY

As discussed in Chapter 1, current climatic conditions are moving beyond historical baselines and will continue to do so even if we can curb greenhouse gas emissions (IPCC 2023). Climate change can have negative, positive or neutral impacts on individual organisms depending on how they respond to the novel conditions brought about by climate change (Chapter 2). A range of climate adaptation decision-support tools exist to aid natural resource managers (hereafter “managers”) in making effective and efficient decisions that consider impacts across populations, species, and landscape scales. Climate Change Vulnerability Assessments (CCVAs) are often a first step in the decision-making process by characterizing which species are most vulnerable, where, and why. Results inform which species or habitats to
prioritize and allocate limited resources based on overall risk level and the timeframe of projected climate changes.

B) COMPONENTS OF CLIMATE CHANGE VULNERABILITY

A well-known definition of climate change vulnerability originates from the IPCC (2007) to describe: “The degree to which biodiversity is susceptible to or unable to cope with the adverse effects of climate change. Vulnerability is a function of the character, magnitude, and rate of climate change and variation to which the system is exposed, its sensitivity, and its adaptive capacity (IPCC, 2007).” Exposure is an external environmental factor (e.g., rate and magnitude of warming), while sensitivity and adaptive capacity are considered intrinsic factors that mediate a species’ overall climate change vulnerability. Similarly, CCVAs measure exposure as the rate and magnitude of change in climatic factors that a species or habitat will (or has) experience(d). Sensitivity refers to how tightly coupled the species’ fitness is to such change and is often characterized as the dose-response relationship between species and specific exposure factors. Adaptive capacity represents the ability of the species (or population, subspecies, etc.) to cope with or adjust to such change through genetic, behavioral, or distributional changes (Beever et al., 2016; Dawson et al., 2011; Thurman et al., 2020).

Table 3.1. Components of climate vulnerability and examples of each.

<table>
<thead>
<tr>
<th>Component of climate vulnerability</th>
<th>Example</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Exposure</strong> describes the nature, magnitude, and rate of climatic and associated environmental changes experienced by a species (Dawson et al., 2011; Foden et al., 2013; Stein et al., 2013).</td>
<td>White et al. (2022) estimated the loss of coastal forested wetlands on the North Atlantic Coastal Plain. The authors found similar rates of loss between unprotected and protected areas, indicating the primary drivers for this loss were due to sea level rise. They estimated that coastal forest wetland loss occurred at a rate of 684 km$^2$ annually from 1996-2016.</td>
</tr>
</tbody>
</table>
**Sensitivity** is the *responsiveness* of a system, either adversely or beneficially, to climate variability or change (IPCC, 2007, 2014).

Laboratory experiments conducted by Holan et al. (2018) found increased water temperature and decreased salinity due to climate change increased the sensitivity of subarctic marine invertebrates to copper contamination by compromising their ability to attach to substrates and affecting mortality rates.

**Adaptive capacity** is the potential, capability, or inherent ability of a species, ecosystem, or human system to cope with or *adjust* to climate change, moderate potential damage, take advantage of opportunities, or adapt to the consequences (IPCC, 2007, 2014).

The breeding season for caribou in the northern Arctic is tightly coupled with seasonal environmental conditions, which are shifting rapidly due to warming. Davidson et al. (2020) found that some populations are adjusting their reproductive phenology, having offspring earlier in the spring to coincide with higher vegetation productivity.

Vulnerability cannot be directly measured (Wade, Hand, Kovach, Muhlfeld, et al., 2017). Rather, exposure, sensitivity, and adaptive capacity are evaluated for each species, population or habitat across a specific geography and timeframe (Figure 3.1).
Pine barren habitats (a) and their associated species, including the eastern whip-poor-will (b) and the ghost tiger beetle (c), are vulnerable to climate change. Here, we give species-specific examples of the three elements of climate change vulnerability—exposure, sensitivity, and adaptive capacity—for this system. In the first panel, the pine barren ecosystem is exposed to climate change in the form of rising temperatures and heavier precipitation events concentrated in shorter amounts of time, manifesting as longer periods of drought interspersed by more intense storms. These factors influence secondary effects of climate change, including higher-intensity wildfires and expanded range of southern pine beetle. The second panel depicts how these impacts might affect the pine barrens habitat and species, or their sensitivity. An example of adaptive capacity is given in the third panel to illustrate the system’s intrinsic ability, or weakness, to adjust to these changes. Finally, we illustrate potential management actions, informed by at least one component of climate vulnerability for each habitat and species. We provide one possible management action—prescribed burning—and protected areas for each example, but many others are possible, as discussed in chapter 4.

Credit: Figure designed by Kate Malpeli, USGS National Climate Adaptation Science Center. Icons courtesy of Tracey Saxby, Sidney Anderson, Catherine Collier, Jane Thomas, Jane Hawkey, Kim Kraeer, and Lucy Van Essen-Fishman, Integration and Application Network (ian.umces.edu/media-library).

Figure 3.1. Examples of exposure, sensitivity, and adaptive capacity for a pine barren habitat and associated species. Figure designed by Kate Malpeli, U.S. Geological Survey.

Recent advancements in predicting potential climate exposure, and knowledge of species sensitivity to recent climate change are detailed in Chapters 1 and 2, respectively; here, we detail
new developments and remaining challenges to understanding species adaptive capacity to climate change and highlight applications for natural resource management.

Adaptive capacity broadly encompasses genetic, phenotypic, or distributional changes (Beever et al., 2016; Dawson et al., 2011; Nicotra et al., 2015) that support a species’ ability to persist-in-place and/or shift-in space—in response to external pressures (Thurman et al., 2020). Adaptive capacity also applies to habitats that support species of concern. For example, different forest types may shift their distribution in space over time (Brandt et al., 2017) while more stationary habitat forming organisms like corals may exhibit phenotypic plasticity (Farr et al., 2021). The expression of a species’ adaptive capacity can be constrained by extrinsic factors (e.g., land use change, competing species). For example, a fish species in a warming stream may exhibit adaptive capacity by shifting its distribution upstream to spawn in colder waters; however, a dam may limit its access to upstream habitats, thereby limiting its ability to behaviorally respond to changing conditions. Thus, managers can enhance adaptive capacity through interventions, such as installing fish ladders, that reduce external constraints or stressors (Beever et al., 2016; Thurman et al., 2022). Adaptive capacity encompasses many life-history characteristics that can be affected by climate-adaptation and management actions (Beever et al., 2016; LeDee et al., 2021; Thurman et al., 2020, 2022). The explicit consideration of adaptive capacity in CCVAs gives managers more points of intervention to help species and habitats threatened by climate stressors.

New tools are emerging to help managers understand adaptive capacity and how to link it to climate-informed actions for species. Thurman et al. (2020) provided a framework for evaluating the adaptive capacity of species or populations based on a set of 36 attributes (e.g., traits, metrics). Attributes are grouped into seven life-history categories: distribution, movement, evolutionary potential, ecological role, abiotic niche, life history, and demography (Figure 3.2). These life-history categories may be more or less relevant to managers based on the taxa being assessed. For example, when assessing a taxa that migrates, migration timing and distance may be more relevant for management interventions than physiological tolerances. Attributes that support a persist-in-place strategy are those that enable the species to survive in situ and can occur through the availability of broad tolerances or existing flexibility (e.g., phenotypic plasticity), or through the acquisition of new traits or expanded tolerances (e.g., evolutionary adaptation). The shift-in-space pathway emphasizes attributes that facilitate tracking of suitable
bioclimatic conditions. Such adjustments in location generally occur in response to changes in one or more limiting environmental variables. We go into greater detail on these pathways and their connection to management in Chapter 4.

Figure 3.2. A framework that utilizes 36 attributes for assessing adaptive capacity (graphically depicted as a wheel), with individual attributes grouped into seven complexes of related characteristics: distribution, movement, evolutionary potential, ecological role, abiotic niche, life history, and demography. Twelve core attributes, representing attributes of particular importance and for which data are widely available, are highlighted in light blue. Figure modified from Thurman et al., 2020 with permission.
C) CLIMATE CHANGE VULNERABILITY ASSESSMENTS

Climate change vulnerability assessments are relatively new decision support tools and differ from other risk assessments in that they explicitly consider and isolate the direct and indirect threats from climate change (Figure 3.3). The assessed relative vulnerability of species or habitats can be used to determine management goals and priorities, enable more efficient allocation of resources, and inform subsequent adaptation strategies (Foden et al., 2019; Glick et al., 2011; Wilson et al., 2020). Conservation biologists developed CCVAs in the 1990s, drawing from risk assessments in other fields, such as natural hazard and disaster planning and efforts to conserve endangered species (Foden et al., 2019). Each CCVA’s analysis addresses location-specific management questions (e.g., state-level, protected area). For multi-species or habitat-based assessments, CCVAs can be used to identify and rank the species or habitats most vulnerable to key climate change factors, and associated impacts (Foden & Young, 2016). Often these ranks are qualitative, e.g., low, moderate, or high vulnerability, but some researchers assess vulnerability on a continuous scale. Because vulnerability is operationalized based on the needs of the assessment, multi-species CCVA results rank relative vulnerability—a comparison of how vulnerable one species is with respect to another—rather than absolute vulnerability—a species’ inherent vulnerability. For example, assessing the vulnerability of all species in a protected area would help inform management planning for that specific area, but lacks broader context and offers only a localized snapshot of vulnerability.

Managers can use these rankings to decide how to triage priorities and allocate limited resources (Foden et al., 2019; Glick et al., 2011; Timberlake & Schultz, 2019; Wade, Hand, Kovach, Muhlfeld, et al., 2017). For example, Colorado Parks & Wildlife used results from a CCVA to inform how their Species of Greatest Conservation Need (SGCN) were binned into tiers, reflecting the relative degree of conservation priority. Although CCVAs do not prescribe management actions, they are an intermediate step in management planning because they provide a framework for prioritizing targets (habitats and species) and organizing potential planning actions based on goals (Galbraith & Morelli, 2017).
D) TYPES OF CLIMATE CHANGE VULNERABILITY ASSESSMENTS

There are multiple approaches to conducting a CCVA, and the process can include one or more elements, such as modeling, expert opinion, literature review and cultural considerations—institutional and regional (Foden & Young, 2016). Understanding the underlying methods in individual CCVAs is essential when comparing results across different assessments. Below, we break down several common approaches and compare them in Table 3.2. These methods are also commonly merged together. For more detailed exploration of each of these methods and further resources, such as databases for life history traits, see Foden et al. 2016 and 2018.

**Expert-elicited opinions** or **stakeholder involvement** can be used in any CCVA process, but they play a more prominent role in some CCVAs. In any CCVA, experts, such as wildlife biologists or resource managers, can advise on the appropriate parameters of a CCVA (Timberlake & Schultz, 2019). For example, Hare et al. (2016) assembled a team of climate and taxa experts, stock assessment and fisheries scientists, ecologists, and oceanographers to determine relevant climate exposure factors and score species’ traits. Timberlake and Schultz (2019) conducted a review and interviewed forest managers to compile stakeholders’ experiences with CCVAs. They found expert opinion or community involvement in the CCVA process helps ensure stakeholder preferences and values are reflected, which improves outcomes since the stakeholders are often the ones carrying out subsequent actions (Timberlake & Schultz, 2019). For example, sometimes climate scientists provide data to managers at a scale that is not actionable; however, by including stakeholder opinions, CCVA authors provide climate data more closely aligned with management activities (Enquist et al., 2017). Additionally, experts can play a larger role in trait-based CCVAs, where they not only determine the parameters of a CCVA but also provide opinions on the species’ vulnerability, as discussed below.

The **correlative approach** to CCVAs focuses on species’ or habitats’ past and current distributions, comparing them to projected future conditions under continued climate change impacts (Pearson, 2009). In doing so, managers can identify areas to prioritize for conservation based on their predicted suitability for species/habitats of concern in the future. Correlative models can be applied across a wide range of taxa at various spatial scales and are often quick and inexpensive to apply. However, correlative approaches can be less useful in assessments of data-poor (understudied) species and can be highly context-dependent. The correlative approach
to CCVAs is spatially explicit, encompassing approaches such as species distribution and climate envelope modeling (Foden et al. 2018).

Although correlative techniques help assess vulnerability and simultaneously identify areas for management action at the landscape level, they do not often consider other non-climate factors affecting species/habitat distributions (Foden et al. 2016). One of the main assumptions of these models is that the species currently resides in all the places it can based on its climatic tolerances (Early & Sax, 2014; Sax et al., 2013). However, due to many other range-limiting factors, including interactions with other species (e.g., competition, symbiotic relationships, pathogens and parasites), availability of resources (e.g., food, water, places to breed), habitat degradation (e.g., due to pollution or land use change), and dispersal restraints, the assumptions of these models are often not met, and these models likely underestimate the climatic niche of the species in question and, subsequently, predicted future ranges (Jankowski et al., 2013; MacPherson et al., 2021).

Broader trends in land use and land cover change may disproportionately impact the distribution of vulnerable populations and communities (Thompson et al., 2019) and when considered along with direct climate impacts provide a more complete picture of vulnerability. Model inputs such as how climate data are averaged/interpolated across space, quality of the species occurrence data, and the variables used to determine climatic suitability thresholds can also significantly impact results (De Los Ríos et al., 2018; Foden et al., 2019; MacPherson et al., 2021). Correlative models also do not take potential adaptation and behavioral shifts that may mitigate the worst effects of changes in temperature and precipitation (Blaisdell et al., 2021; Ducatez et al., 2020). Additionally, models are based off historical climate patterns that may not hold true as climate change pushes systems beyond historical bounds and enters a non-analogue future (Scherrer et al. 2021). Given all model assumptions and data limitations summarized above, managers should take great care when interpreting results of these models (and any model used to inform management decisions generally [Grimm et al. 2020]). Despite the limitations to correlative climate niche models, for many species, these approaches remain the most accessible way to predict future range shifts due to climate change (Tingley et al. 2014).

The mechanistic approach to CCVAs focuses on the processes by which climate change impacts species/habitats of interest (Foden & Young, 2016). It can also include interactions between climate change and other stressors. Such mechanisms include changes in habitat and
resource availability, affecting species’ ability to persist in their current range or population size. For example, changes in water temperature determine the sex ratio of turtle eggs, which can have major impacts on future reproductive success and population growth (Santidrián Tomillo & Spotila, 2020). Mechanistic models, although capable of important insights into underlying eco-evolutionary processes, can be difficult to build because physiological data about many species of interest are sparse, making this approach more costly to implement than others (Foden et al. 2016).

The **trait-based** (or index-based) approach focuses on connections between climate impacts and species’ intrinsic characteristics, explicitly assessing their sensitivity and/or adaptive capacity. Combined with a measure of exposure, trait-based CCVAs rank species on their overall vulnerability (Foden et al., 2019). Trait-based approaches allow for rapid assessments of multiple species and are simpler to implement than mechanistic approaches. Such approaches are also useful for ranking, categorizing, and identifying thresholds based on the suite of characteristics used in the assessment, and can use information derived from both correlative and mechanistic assessments (e.g., Powell et al., 2017). However, the exact vulnerability thresholds of selected traits may not be known without a mechanism-based understanding of species-climate relationships. As discussed above, these CCVAs often rely on expert opinion to identify and generate a list of traits used in the ranking process. The adaptive capacity framework described previously is an example of a trait-based approach that utilizes expert elicitation and includes a metric for evaluating the strength of evidence (Thurman et al., 2020). When reviewing a trait-based CCVA, scrutinizing these more qualitative determinations of vulnerability is crucial to interpreting species scores or ranks (Foden & Young, 2016).

**Trend-based** CCVAs are a subset of the trait-based approach that focus on the demographic traits most closely linked to species decline and vulnerability, such as population size (Pacifici et al. 2015). Directly monitoring the changes in these numbers recognizes which species/populations are fluctuating without elucidating the complex underlying mechanisms (Wheatley et al., 2017). However, the trend-based approach also incorporates some trait data (e.g., thermal limits) to lend context to the observed demographic trends and assess the likelihood that patterns will continue on predicted trajectories (Wheatley et al., 2017). Therefore, users need to be aware of the data used to establish the baseline (e.g., range of years) when interpreting a trend-based CCVA.
Multiple CCVA approaches discussed above can also be combined, harnessing the advantages of each while balancing some of their drawbacks (Foden & Young, 2016). Combining results, however, requires careful consideration of the original study-specific context, methodology, and spatial scope to understand potential outcomes for a species or habitat under a suite of conditions (Wade, Hand, Kovach, Muhlfeld, et al., 2017). In particular, CCVAs often blend mechanistic, correlative, and trait-based approaches to provide a multifaceted view of climate change impacts on species of concern. Trait-based techniques add biological relevance to more quantitative ones, while correlative methods add a spatial component to other methods, helping inform management decisions (Foden & Young, 2016). Expert elicitation may also be incorporated into any CCVA approach, such as to choose the underlying criteria.

No matter the approach, Wade et al. (2017) argue it is best practice for authors of CCVAs to include a conceptual (or heuristic) model when creating and reporting results. When reviewing a CCVA, a conceptual model outlines how a researcher thinks their proxy metrics represent the elements of vulnerability and a visual representation of how the ecological system functions (Haynes, 2019; Wade, Hand, Kovach, Muhlfeld, et al., 2017). This element creates a clear hypothesis for how a system works and can help identify data gaps (Wade, Hand, Kovach, Luikart, et al., 2017). In addition, because vulnerability can be operationalized in different ways, conceptual models facilitate comparing CCVAs with a clear understanding of what each includes (Haynes, 2019; Wade, Hand, Kovach, Muhlfeld, et al., 2017). For example, Nyboer et al. (2021) built a flow chart for their CCVA methodology, showing how different system inputs fit together and the data sources used to calculate these inputs. Specifically, this chart allowed for a clear understanding of the non-climate factors included in their approach, such as socioeconomic value of the species and conservation effort (Nyboer et al., 2021). Conceptual models can also help clarify the differences in vulnerability between two species inhabiting the same habitat and range whose climate exposure is similar but differ in their adaptive capacity, sensitivity or other complex issues (Wade, Hand, Kovach, Muhlfeld, et al., 2017).
**Table 3.2.** Pros and cons of common CCVA approaches.

<table>
<thead>
<tr>
<th>CCVA approach</th>
<th>Pros</th>
<th>Cons</th>
</tr>
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| Expert elicitation | - Captures expert knowledge which may not be otherwise represented in scientific literature  
- Aligns CCVA with stakeholder values | - CCVA results reflect the strengths and limitations of the chosen experts consulted  
- Trait rankings/categories can vary widely in their scoring |
| Trait-based | - More qualitative approach that requires less technical “know-how” and harnesses existing knowledge to compare species’ vulnerability  
- Often faster | - May be less accurate, especially when data gaps are present or links between climate impacts and species traits are unclear |
| Correlative | - Relatively speedy way to quantify and identify potential future distributions of species/habitats in relation to current ones  
- Can inform management priorities at the landscape scale | - Algorithms used may produce different outcomes at different spatial scales  
- Do not account for non-climate causes of distribution change or limitation (e.g., land development; invasive species)  
- Limited by understanding of a species’ fundamental versus realized niche  
- Limited by historical climate patterns which may not be applicable in a non-analog climate future |
<p>| Mechanistic | - Data-backed, focusing on actual | - Often more costly and time- |</p>
<table>
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<th>processes by which climate change impacts species/habitats</th>
<th>consuming due to the need for empirical support</th>
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<tr>
<td>- Limited by understanding of a species' life history, and may not be possible for all species, e.g., cryptic or understudied species</td>
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| Trend-based | - Focuses directly on the factors of concern for a species at risk (e.g., population size) | - Incomplete knowledge of these demographic trends, and/or of the likelihood they will continue, limits this approach’s utility |

**i. Uncertainty is a part of every CCVA**

CCVAs offer valuable insights as to how projected climate change is expected to impact species and ecosystems but contain multiple sources of uncertainty. These include, but are not limited to, the range of possible futures (e.g., high-emissions vs low-emissions climate scenario) and timelines chosen to predict system responses, incomplete knowledge of the species or habitat assessed, and scope of the studies compiled to make determinations. To understand process-based uncertainty, it is important to examine how authors identify and report the type and strength of evidence used in a CCVA. This can be as simple as reviewing how many sources (e.g., published or number of experts) were used to generate a ranking and determine the level of confidence in an outcome. In some studies, experts are asked to both provide an overall ranking and report their confidence level in that ranking based on available evidence. For example, Galbraith & Morelli (2017) had experts rate how traits may be related to vulnerability and their uncertainty in these rankings. Similarly, Thurman et al (2022) provide criteria for evaluating the availability, quality, and consistency of evidence used in their trait-based assessment. While managers are accustomed to acting in the face of uncertainty, some of the less familiar types of uncertainty stemming from climate data may inhibit decision-making. In such cases, additional decision support tools and approaches may be useful for characterizing unknowns and informing next steps (see Chapter 4).
ii. Common CCVA tools

The Climate Change Vulnerability Index (CCVI) is an expert-elicitation trait-based tool developed to assess species vulnerability in North America (Foden et al., 2019; Young et al., 2012; Young et al., 2016). The CCVI contains scoring elements that collectively reflect factors known to influence climate change vulnerability and uses this information to place species into broad categories of vulnerability. The tool is designed to help resource managers screen relatively large numbers of species from diverse taxonomic groups to identify the species most vulnerable to climate change and the factors leading to that vulnerability.

The CCVI is one of the most commonly used approaches to assess fish and wildlife across the Northeast region, and results for 7 applications are summarized in Appendix 3. The CCVI has been updated several times since it was first released in 2010 (Young et al., 2011, 2015), and as of 2023, is in version 3.02, with a new version currently in progress. When interpreting CCVI results (or any other CCVA method), it is important to note that each tool has assumptions incorporated into the algorithm that generates results (Wheatley et al., 2017). For example, some CCVI users reported they had difficulty interpreting results because they could not access the internal algorithm (Young et al., 2015). Currently, a joint effort by the U.S. Geological Survey (USGS) and NatureServe is working to overhaul the CCVI tool to include changes to the interface, updated climate data, expanded functionality, and other changes.

A complementary approach for assessing habitats is NatureServe’s Habitat/Ecosystem CCVI (HCCVI). Like the species CCVI framework, the HCCVI incorporates climate trends, trait data, and expert knowledge to quantify the exposure, sensitivity, and adaptive capacity of ecosystems (Comer et al., 2019). This framework has been applied primarily in the western U.S. to date, although its approach is transferable to eastern ecosystems (Comer et al., 2022).

The Ecosystem Vulnerability Assessment Approach (EVAA), developed by the Northern Institute of Applied Climate Science (NIACS), is a CCVA approach used to assess forest habitats in the Midwest and eastern U.S. The EVAA uses a combination of climate modeling and expert-elicited opinions to categorize the climate vulnerability of various forest types/ecosystems (Brandt et al., 2017). It also harnesses expert knowledge to discuss the forest management implications of each vulnerability assessment (Brandt et al., 2017).
decade, the EVAA has been used to complete several comprehensive regional forest vulnerability assessments, covering the New England, Mid-Atlantic, and Central Appalachian regions in the Northeast, as reported in Appendix 3 (Butler et al., 2015; Butler-Leopold et al., 2018; Janowiak et al., 2018).

**iii. New developments in CCVAs**

Many CCVAs published since the last iteration of the State Wildlife Action Plans in 2015 (e.g., De Los Ríos et al., 2018; Foden et al., 2019; Wade, Hand, Kovach, Luikart, et al., 2017) have extended the field in new directions. These new developments include widening the scope of CCVAs to assess the global state of a taxon and conducting CCVAs for previously unassessed species (Böhm et al., 2016; Hossain et al., 2018). A growing number of CCVAs now exist for coastal and marine species and habitats (Allyn et al., 2020; Farr et al., 2021; Hare et al., 2016; Lettrich, 2020; Morrison et al., 2015) whereas prior to 2015, few marine species had been assessed using CCVA approaches. There are also a growing number of global-scale CCVAs that provide a broader context, and spark ideas for management and collaboration that consider widespread climate pressures a species, taxon or system is facing worldwide (Bohm et al. 2016, Hossain et al. 2018). Although global CCVAs may include less locally specific information for management planning, they provide important context for interpreting state or regional vulnerability in the context of global distribution of a species or habitat. As the CCVA field continues to expand, platforms such as the Climate Change Research Updates e-mail listserv (Jason_Goldberg@fws.gov) and the Climate Toolbox (https://climatetoolbox.org/) are sources to look for newly released studies focused on natural resources. The CRAVe database which previously served as a clearing house for CCVAs is now housed in the Climate Adaptation Knowledge Exchange (CAKE) (https://www.ecoadapt.org/programs/knowledge-exchange). Although this database is not expected to be updated, it is a repository of past CCVAs and allows users to see what species and habitats have already been assessed.

In addition to the broader scope of CCVA subjects, species’ and habitats’ vulnerability is being conceptualized in more complex ways that capture risk across migratory corridors and seasonal habitats. For example, migratory species such as the American Kestrel (*Falco sparverius*) have been assessed by multiple CCVAs to create a more holistic look at vulnerability throughout their full life history (Culp et al., 2017; Galbraith et al., 2014). Combining two or
more established frameworks can lead to new insights and a more comprehensive understanding of a species’ overall vulnerability. One such CCVA (Farr et al., 2021), which focused on aquatic habitats in the Northeast U.S., combined an approach developed for the Northeast Association of Fish & Wildlife Agencies for a habitat assessment with NOAA’s Fish Stock Climate Vulnerability Assessment framework (Morrison et al., 2015). Other methods integrate landscape variables and species traits to more realistically model species vulnerability in situ (Garcia et al., 2014; Nadeau & Fuller, 2016; Pearson, 2009). Assessments such as the Massachusetts Rapid Assessment Protocol (MRAP) evaluated both climate and non-climate vulnerabilities, as well as their associated uncertainty and causal factors (Galbraith & Morelli, 2017).

Researchers are also including non-climate anthropogenic forces to expand the parameters of direct and indirect risk exposure a species might be facing (Denryter & Fischer, 2022; Dudley et al., 2021; Segan et al., 2015). An evaluation of northern forest ecosystems incorporated the effects of non-native insect pests, such as the Hemlock Wooly Adelgid (Adelges tsugae), which was initially introduced to the Northeast U.S. due to human activity, into its vulnerability assessments (Janowiak et al., 2018). To add further complexity, the Hemlock Woolly Adelgid is also expanding its invasive range northward due to warming winters; this growing threat encapsulates both an indirect effect of climate change and an anthropogenic factor in forest habitat vulnerability.

Other CCVAs have included human communities and their ability to respond to climate change as a measure of adaptive capacity. For example, one CCVA included state-level research and funding capabilities, monitoring stations, and number of academic institutions to inform the societal-adaptive capacity for 112 estuarine watersheds in five regions across the continental United States, and illustrated this definition in an interactive web tool (Montefiore et al., 2023). These studies fall under the broader category of socio-ecological assessments, which focus on the relationship between, and the linked vulnerability of, ecological and human systems. Such approaches are especially relevant when human needs and wellbeing are closely tied to local ecological systems, such as in forestry and fishing communities, and the actions of social systems can either threaten or bolster ecological systems. One such study assessed the forestry industry in Maine at a county level, including climate factors affecting tree species as exposure and forest business, infrastructure, and market-related factors in the measure of sensitivity (Soucy et al., 2022). In another example, Dudley et al. (2021) created a framework for assessing
vulnerabilities in species-specific fisheries that includes measures of adaptive capacity for the fishing community.

Another system of socio-ecological assessment is the Climate Vulnerability Index (CVI). The CVI was originally developed to assess the vulnerability of World Heritage properties (Day et al., 2020) but has the potential to be modified to assess other fixed areas of socio ecological importance such as National Wildlife Refuges (NWR), National Parks, Marine Protected Areas (MPAs), and Tribal resources. The CVI advances the field of CCVAs by creating a methodology that assesses the vulnerability of the economic, social and cultural connections communities have with World Heritage sites. Similar to other recent socio-ecological CCVA approaches (e.g., Montefiore et al., 2023), it incorporates the associated communities’ adaptive capacity into the final vulnerability score (Day et al., 2020). An important and somewhat unique aspect of this framework is that it emphasizes a collaborative, community-based process that specifically engages and incorporates the historical and traditional knowledge and observations of the people and communities surrounding the site (Day et al., 2020). This differs from many other CCVA methodologies, which derive results from small groups of selected scientific experts or resource managers. Notably, the methodology was developed to better fit the needs of the Traditional Owners of the land and sea country (e.g., Archer Point, North Queensland, Yuku Baja Muliku) and incorporate their values into the assessment process (Hale et al., 2022).

The CVI methodology is currently being expanded to assess other types of protected areas and offers a promising method to engage community members into the assessment process with the added benefit of gaining trust and social investment in resulting climate adaptation and management outcomes. One caveat for transferring the methodology to other protected areas is that World Heritage properties have explicit statements of value to justify why they were created. These statements may correlate to organizational mission statements or may need to be developed from scratch depending on the application.

CCVAs conducted by Tribal nations represent a new pathway for integrating multiple types of knowledge into the CCVA format. Cultural and social elements are key aspects of Tribal CCVAs and include insights from Tribal members gleaned from long-standing relationships with nonhuman beings. The Great Lakes Indian Fish and Wildlife Committee (GLIFWC Climate Change Team, 2023) conducted a CCVA of 60 beings/species that were “of interest to GLIFWC’s member tribes across the Ceded Territories” and blended Traditional Ecological
Knowledge (TEK) with Scientific Ecological Knowledge (SEK) in the results (GLIFWC Climate Change Team, 2023). The authors of the report conducted interviews with “Tribal elders, harvesters, and knowledge holders from each of the GLIFWC tribes” to gather the cultural importance of beings and understand environmental effects over time (GLIFWC Climate Change Team, 2023). For example, the decline of Waabooz (Snowshoe Hare, *Lepus americanus*) is demonstrated by interviewees saying they have noticed fewer Waabooz footprints in the snow, and they worry that the cultural stories of Waabooz will live only in memories because younger generations will never see one (GLIFWC Climate Change Team, 2023). The authors also used the CCVI tool to assess vulnerability of beings to future climate change, such as how projected warming and resulting lack of snow will affect Waabooz distributions in the future (GLIFWC Climate Change Team, 2023). A number of Northeast Tribal nations have participated in CCVA writing workshops hosted by United South and Eastern Tribes (USET) staff in recent years. Final results can be found on the USET website (https://www.usetinc.org/library/videos-reports-and-publications/) or at https://tribalclimateguide.uoregon.edu/adaptation-plans when they are completed.

iv. Other relevant management and risk frameworks

While CCVAs focus on threats due to climate change, several other emerging management and risk frameworks can provide valuable information about the comprehensive and interactive risks from multiple climate and non-climate stressors facing natural resources in an era of global change.

Climate change is rarely, if ever, the sole threat a vulnerable species or habitat faces (TCI & NFWDTC, 2023). Invasive species and land-use change are widely considered the most urgent contemporary threats to biodiversity in the Northeast, however, both of these threats are amplified by climate change (TCI & NFWDTC, 2023) and recent studies argue that current biodiversity loss on a continental scale has to date, primarily been driven by factors other than climate change (Caro et al., 2022; Waldman & Quinn, 2022). This is especially true in mountain and coastal areas that have relatively more thermal refugia and buffers compared to southern regions and other parts of the U.S. While CCVAs have increased our explicit understanding of climate risks, consideration of climate threats in isolation may underestimate a species’ absolute vulnerability, especially for near-term (e.g., 2050) timelines. As the field continues to move
towards implementation of climate adaptation practices, integration of climate with other, more immediate threats can help managers understand species’ overall vulnerability across spatial and temporal scales.

**Cumulative risk assessments** incorporate multiple threats or drivers of vulnerability and many directly link their findings to relevant conservation and management practices. Recent applications have been used to characterize threats and identify data gaps around offshore wind energy development (e.g., Popper et al., 2022), to develop the “State of the Ecosystem” reports for regional Fishery Management Councils along the Northeast and mid-Atlantic coast (Northeast Fisheries Science Center (U.S.), 2023b, 2023a). Cumulative risk assessments also consider the interactive effects of climate, pests, and weeds in forest ecosystems (Fisichelli et al., 2014) and fishing, climate, and pollution in marine systems (Butt et al., 2022). See Box 3.1 for additional examples.

**Ecosystem-based management** is a comprehensive assessment of all social, ecological and economic uses of a defined system and the potential levers of management that affect community structure and function (Link et al. 2011; Patrick and Link 2015). For example, in marine environments, fishing is often the focus because it has been the strongest influence on ecosystem functioning to date (Link, 2010). More recently climate change is increasingly being incorporated into models that evaluate population dynamics and harvest limits (Wiese & Nelson, 2022). This approach explicitly combines management with vulnerability/risk assessment, and it can provide valuable insight into species’ and habitats’ climate vulnerabilities by taking a wider view of the system. For example, Gaichas et al. (2018) documented an ecosystem-level risk assessment conducted by the U.S. Mid-Atlantic Fishery Management Council that included five elements of risk and associated management goals with each. Another recent study considered diadromous fish communities in the context of their marine, estuarine, and freshwater ecosystems, taking into account the ecosystem services each provides to inform management decisions (Ouellet et al., 2022).

**Response-focused assessments** quantify species’ vulnerability, or individual components of it, in ways that diverge from the stricter CCVA approach. Some of these focus on specific factors affecting one of the components of vulnerability, such as a study quantifying a species’ “evolutionary potential,” a facet of adaptive capacity (Forester et al., 2022; Thompson et al., 2023). Others eschew the standard definition of vulnerability (exposure, sensitivity, and adaptive
capacity) altogether (Fortini & Schubert, 2017). Often, these assessments more explicitly incorporate the literature on biological responses to climate change (Chapter 2), as in the case of a review of animals’ endocrine responses to anthropogenic factors including climate change (Seebacher, 2022).

**Positive response-focused assessments** center on the species *increasing* their populations or ranges due to climate change and their associated ecosystem impacts; these can often take similar forms to other risk assessments. To date, positive-response assessments have focused on non-native invasive species, pests, and opportunistic generalists. Both native and non-native species at their northern range limit in the region are more likely to see beneficial population effects of global warming (Wallingford et al., 2020). A recent framework by Wallingford et al. (2020) laid out an approach for modifying an invasive species risk framework to assess the impacts of range-shifting native and non-native species, which was then applied in North American coastal marine ecosystems by Henry and Sorte (2022). The authors recommended pairing results with vulnerability assessments to identify species that need active management and the study provided an example of how multiple frameworks can be coupled to develop a more complete picture of climate change and other impacts on biodiversity (Henry and Sorte 2022). The Regional Invasive Species and Climate Change (RISCC) management network has coproduced research results like this with natural resource managers and practitioners to effect actionable science.

**Box 3.1.** Renewable energy infrastructure impacts on RSGCN and their habitats, and associated resources including cumulative risk assessments.

Hydroelectric, solar, and wind energy infrastructure are each associated with potential impacts to natural systems. Because one of the primary factors driving their expansion is the goal to decrease reliance on fossil fuels, the effects of renewable energy on ecosystems can be considered a secondary effect or maladaptive of climate change itself (Stein et al., 2013). Although a full analysis of renewable energy impacts on ecosystems is beyond the scope of this report, selected regionally relevant examples and resources about the impacts of these technologies on RSGCN and their habitats are summarized below.
**Hydroelectric power**

Dams generating hydroelectric power have been known for decades to disrupt fish migration and movement and impede access to spawning grounds. As a result, many mitigation measures, including fish ladders and lifts, have been installed to facilitate passage for RSGCN such as Atlantic Salmon (*Salmo salar*) (Cada and Sale 1993), River Herring (*Alosa pseudoharengus*), American Shad (*Alosa sapidissima*) and other diadromous fishes (Hare et al., 2016; Waldman & Quinn, 2022). Hydropower infrastructure also impacts terrestrial habitats and species, such as flooding land for dam reservoirs. Dam removal in New England has led to increased aquatic habitat connectivity, re-linking 3,770 km of waterways by one estimate (Magilligan et al., 2016).

**Solar power**

Solar photovoltaic (PV) systems are more prevalent in the western United States but are increasingly implemented in the Northeast. Many solar energy risk assessments do not assess impacts to wildlife or habitats (e.g., Qin, 2022); however, a recent report from the U.S. Department of Energy summarized solar impacts on wildlife, including RSGCN such as the American Woodcock (*Scolopax minor*) and Loggerhead Shrike (*Lanius ludovicianus*) and habitats such as oak woodlands and vernal pools (2021). Several Northeast states have also developed their own guidance for solar energy projects to take natural resources such as RSGCN into account (Maine Department of Inland Fisheries and Wildlife, 2020), from project siting to mitigation, as have larger-scale organizations like the IUCN (Bennun et al., 2021).

A major concern with PV infrastructure is its disruption to existing habitat. Mass Audubon estimates that up to a quarter of land development in Massachusetts from 2012 to 2017 is attributed to ground-based PV installation (Ricci et al., 2020). In Rhode Island, solar development is the cause of two-thirds of forested land cleared between 2018 and 2021 (Rudin, 2022). The impacts of this development on RSGCN are not well-documented but highlight a need for further study (U.S. Department of Energy 2021). Although this is not possible with all habitat types (e.g., forests), some research also indicates the utility of co-existing vegetation and PV infrastructure (Choi et al. 2023).
Wind power

Terrestrial wind turbines can pose a threat to birds and bats, including RSGCN, both directly (by blade strikes) and indirectly (by altering their preferred flight paths). One study found that Golden Eagles (*Aquila chrysaetos*) in Appalachian Pennsylvania faced varying degrees of exposure to wind turbines based on the turbines’ locations and the eagles’ flight patterns, and that relocating some of the highest-risk turbines, or strategically placing future turbines in lower-risk areas, would reduce these impacts (Miller et al., 2014).

Offshore wind farms are a quickly growing field of the renewable energy sector (Allison et al., 2019). They raise similar concerns to terrestrial turbines for birds and bats (Allison et al., 2019; True et al., 2021), and multifaceted potential impacts for marine species and habitats (Copping et al., 2015). For example, one recent study modeled overlapping core habitat use by 177 marine taxa in the Northeast U.S. Continental Shelf with the spatial footprint of proposed offshore wind development areas (Friedland et al., 2023). Results found a disproportionate amount of overlap in key seasonal habitat and development sites for a number of RSGCN, SGCN, and regionally important forage fish including Alewife (*Alosa pseudoharengus*), Atlantic Menhaden (*Brevoortia tyrannus*), Atlantic Herring (*Clupea harengus*), Sand Lance (*Ammodytes* sp.) and others. Offshore wind impacts on fish, aquatic invertebrates, marine mammals, sea turtles, birds, and bats are synthesized in workgroup reports from the New York State Energy Research and Development Authority’s State of the Science (SOS) Workshop on Wildlife and Offshore Wind Energy. These workshops happen every 2 years, and the next planned for 2024 intends to provide updates on risks and emerging research around wind energy (Cook et al., 2021; Gitschlag et al., 2021; Hein & Jenkings, 2021; Popper et al., 2021; Southall et al., 2021). The Responsible Offshore Science Alliance ([https://www.rosascience.org/](https://www.rosascience.org/)) is another cross-disciplinary research group studying the intersection of offshore wind energy and fisheries management.

The USGS Energy and Wildlife research program provides regular updates on relevant publications and projects to this topic.
v. Limitations and future directions of CCVAs

CCVAs are still relatively new conservation tools and are constantly being updated to incorporate new factors and expand methodologies to better address system responses to climate and other threats. CCVA results can be viewed as hypotheses for how a species will be impacted by climate change in the future and thus need to periodically reevaluate conclusions to determine their predictive power and long-term effectiveness (Wheatley et al., 2017). There are several known limitations to CCVA approaches which have been discussed throughout this chapter. Below are four key considerations to keep in mind when using CCVA results.

a. CCVAs rarely include indirect or cascading effects

Climate change may cause indirect or cascading effects that are not captured in CCVAs (Cunningham et al., 2021; Maxwell et al., 2015; Segan et al., 2015), and this can lead to over- or underestimating vulnerability scores. Species that are labeled “presumed stable” may still face climate threats, but those threats just do not reach specified thresholds (Sneddon & Hammerson, 2017) or the tipping point that would cause a non-linear effect is not known (Foden et al., 2019). Indirect effects can also include species linked by trophic interactions through predator-prey, competitive or host-parasitic relationships.

In some cases, changes to trophic interactions can limit species’ range expansion, phenological shifts, and survival more than direct exposure to the effects of climate change (Staudinger et al., 2021). For example, a bird species may have behavioral adaptations that allow it to exploit cooler patches during heat events, which could be defined as high adaptive capacity and low exposure to temperature changes; however, using those patches may limit its ability to find food depending on the response of its prey (Cunningham et al., 2021). These complex dynamics may not be fully understood or captured in a CCVA’s conceptual model. Other cascading or indirect factors like disease spirals, invasive species, ecosystem changes as “native species” move to new areas and others will affect vulnerability. Additional tools like scenario planning, priority threat management, and horizon scanning could help incorporate these effects and will be discussed further in Chapter 4.

Individuals and communities—both ecological and human—may adapt to climate change impacts, and these adaptations could indirectly exacerbate the vulnerability of RSGCN (Maxwell et al., 2015). Segan et al. (2015) conducted one of the few species-focused CCVAs that
considered the indirect effects of climate change and potential anthropogenic impacts as the human community shifted. They found that a subset of species in an Important Bird and Biodiversity Area in Southern Africa would not be impacted by climate change initially but would be as the local human communities adapted to climate change (e.g., changes in agricultural footprints). Similarly, other studies have predicted increases in human-wildlife conflicts as species and humans shift and adapt (Abrahms et al., 2019; Silber & Adams, 2019).

Cascading effects could take many other forms. Sneddon and Hammerson (2017) note that in their CCVA moose were ranked as “presumed stable.” They caution that this ranking does not take into account host-parasite interactions where parasites will likely increase as winters warm, and “presumed stable” is likely an underestimate of moose vulnerability. It is also possible CCVAs overestimate vulnerability, as in the case where a new species could colonize an area and have positive synergistic effects with the historical species. These examples point to species-specific actions for research and monitoring to fill gaps as part of subsequent adaptation plans.

b. Some species and habitats do not have enough data to be part of a CCVA

In some cases, not enough is known about species and habitats to evaluate them in a CCVA. Several authors have called attention to the need for more information on biological traits. This includes the need for more occurrence data, species ranges, better treatment of interspecific interactions, the incorporation of molecular information, adaptive genetic change and phenotypic plasticity, and extreme climate and ecological events (e.g., extreme cold that kills pests) (De Los Ríos et al., 2018; Foden et al., 2019; Griffis-Kyle et al., 2021; Thurman et al., 2020). This information may be particularly important for species with naturally small ranges, or when gathering information on true estimates of ranges for species that have contracted ranges due to anthropogenic pressures.

c. CCVAs inform decisions but are only a first step in the overall process

After conducting a CCVA, managers have reported that it can still be difficult to use results in decision-making (Timberlake & Schultz, 2019; Young et al., 2015). CCVA results are subsets of data that can help managers make decisions, but those decisions are informed by many other factors (e.g., political, social, financial). For instance, a manager with very limited
resources could choose to focus on species that are not vulnerable (i.e., put resources toward species that are most likely to persist through climate change with the least amount of help). Whereas another manager could take the same CCVA results and focus on the most vulnerable species. These two managers are weighing multiple values, limitations, and priorities when making a decision informed by CCVA results.

A few CCVAs and alternative approaches discussed above contain management considerations, and many authors of CCVA papers will give recommendations for next steps in their discussion sections. These are good sources of ideas to facilitate the development of subsequent management actions. Chapter 4 goes into more detail about additional decision-making tools as well as climate adaptation strategies and actions.

d. **CCVAs are subjective**

Finally, CCVAs are subjective because whoever is designing the assessment is picking the variables most salient to them, and so CCVA results may not be applicable for other decision makers (De Los Ríos et al., 2018; Timberlake & Schultz, 2019). Before using CCVA results to inform management actions outside the original scope of the assessment, users need to consider the CCVA’s contextual features. Some of these considerations might be deal breakers, or may just indicate the CCVAs results should be interpreted with caution (for example, use of an older climate model when newer downscaled projections are available). We present these considerations below as an infographic (Figure 3.3) and more detailed text to both inform how the results synthesized in the appendices can be interpreted and applied to specific decisions, and for evaluating any future CCVAs.
Considerations when assessing a CCVA

Figure 3.3. Considerations when assessing a Climate Change Vulnerability Assessment.
a. Geographic scale

The geographic scale of a CCVA can impact the vulnerability scores for species or habitats (De Los Ríos et al., 2018; Foden et al., 2019; Nadeau & Fuller, 2016; Tuberville et al., 2015). Some species- or taxa-focused CCVAs may include the entire range of a species (e.g., Bohm et al., 2016). These scores may be less applicable to a management question focused only on a certain part of a species’ range (for example, the southern edge). If a CCVA is focused on a geographic location, like a protected area, those vulnerability scores may be difficult to interpret outside that area because exposure may be more intense or geographically isolated populations in the area may have limits to their fundamental adaptive capacity (Beever et al., 2016; Foden et al., 2019). In essence, the vulnerability of a population or sub-population can differ from the vulnerability of the species as a whole, or in other parts of the species’ range.

However, both types of geographic scopes can still inform management decisions outside their original context. For example, a CCVA finding that a species is vulnerable throughout its entire range, such as a bird that migrates between North and South America and has many stopover sites, indicates that local management efforts should be coordinated with regional, national, and international partners. Managers can use CCVAs for geographies outside of their jurisdiction to expand their knowledge of a species’ or habitats' vulnerability throughout its range, and communicate with other institutions to ensure that it is preserved in areas of low vulnerability because it may continue to decline elsewhere.

b. Ecological scale

Because CCVAs sometimes assess relative vulnerability it is important to consider which species, taxa, or habitats are included in an assessment. For example, a CCVA looking at songbirds in a specific geography will provide results on which songbirds are highly vulnerable or not. However, even if a songbird is ranked Highly Vulnerable, they may not be the most vulnerable bird in that geographic area, and birds may not be the most vulnerable taxon in the area.

In addition, CCVAs based on generalized life history traits may be missing nuances among populations. For example, different populations within a species can have varying responses to climatic variation (Dalton et al., 2022; Jackson et al., 2022; Legett et al., 2021). One study using a regression model to link environmental factors with life-history traits and
population dynamics argues that studies using life history characteristics from a single population may not capture the full range of responses by that species across populations (Jackson et al., 2022). For CCVAs this could be important in the reverse as well, where the life history traits of the species in general may not be representative of the subset present in the area of interest.

Unlike most animal species, habitats vary widely in how they are named and classified. Thus, the classification system used is an important consideration for CCVAs evaluating habitats and those assessing species vulnerability in different habitats (Anderson et al., 2015). Like geographic scale, the scale of habitat definition can affect the results of a CCVA. For example, a broader category of habitat, such as temperate broadleaf forests, may be less vulnerable overall, but may include specific taxa or species assemblages that are more at risk.

c. How climate change was modeled

Future climate projections are uncertain for several reasons (e.g., Hawkins & Sutton, 2009). Major factors include uncertainties associated with (a) the imperfect representation of processes included in climate models (model uncertainty), (b) the unpredictable nature of climate variability that occurs naturally (internal variability), and (3) not knowing exactly the future trajectory of greenhouse gas emissions (scenario uncertainty). To account for this, often climate models use a suite of different inputs. For example, the climate projections presented in Chapter 1 are based on a suite of climate models and multiple emissions scenarios that bracket the range of plausible future outcomes, ranging from best-case to worst-case, for variables of interest. Similarly, many CCVAs use a suite of climate scenarios, where the difference in results from a low emissions scenario and a high emissions scenario may help convey and assess uncertainty by presenting a range of possible futures. This is because some scenarios may underestimate or overestimate risk, and presenting a range of vulnerability scores can help triage species that may be sensitive to any amount of climate change, versus those who may only be sensitive to a worst-case scenario (Powell et al. 2017). The scale or resolution of input variables used in climate models can also affect the results, especially when examining the impacts at smaller, local scales, an issue that Chapter 1 also addresses.

Different biophysical variables may lead to different analyses of climate exposure. For example, including or excluding variables like temperature, precipitation, drought frequency,
frequency of extreme events, etc. could impact the level of exposure projected for each species. In addition, these results vary by species or habitat; for example, a migratory species only present in the area of interest during half the year will not be directly exposed to climatic changes in the area during the other half of the year. Some species may be strongly affected by changes such as sea level rise, while other terrestrial species will not be directly exposed to it. These multiple sources of variation all shape how a climate model predicts future exposure for species or habitats of interest.

*d. When was the CCVA conducted*

We present results from CCVAs from 2010 to present, but caution that updated information on species’ exposure, sensitivity, or adaptive capacity to climate change may have become available since a study was conducted. CCVA results may also be outdated if a species’ range or abundance has shifted due to climate change or non-climate stressors such as habitat loss due to land use change. Consequently, it is important to consider the time period that each assessment has covered in its information review and revisit the literature for more recent findings.

*e. How vulnerability was defined*

While vulnerability (V) is defined as a function of exposure (E), adaptive capacity (AC) and sensitivity (S), studies may differ in how they conceptualize the relationship between these factors as well as how each component is assessed. For example, some authors conceptualized vulnerability as V=E+S+AC, while others conceptualized it as V=f(E, S, AC) where vulnerability is a function of E, S, AC but do not explain their conceptual relationship between the three factors (Haynes, 2019). This difference in conceptual models could lead to different inputs and different results (Wade, Hand, Kovach, Muhlfeld, et al., 2017; Wheatley et al., 2017). Adaptive capacity is often overlooked or not fully considered in many CCVAs (De Los Ríos et al., 2018; Haynes, 2019), in part because certain species attributes are not well known. In some cases, this has led to the use of sensitivity alone as a proxy for adaptive capacity (De Los Ríos et al., 2018). There have been major strides over the last decade in evaluating adaptive capacity within a vulnerability context (Beever et al., 2016; Thurman et al., 2020) and these advancements provide opportunities for future research.
E) CONCLUSION

Climate change is increasing risk to many species of conservation concern and poses new challenges to managers. Climate change vulnerability assessments are a first step in the climate adaptation planning process and help identify the primary threats, known responses, and uncertainties of species, assemblages, and habitats. These assessments are context dependent, and by cross referencing results from studies with different inputs, the results can paint a complex picture of a species’ or habitats’ vulnerability. Once a species or habitat has been assessed, climate adaptation plans and actions can be developed that draw from the lessons learned in the CCVA process and develop actions that are based on species’ responses to environmental and ecological drivers to help species shift in space or adapt and persist in place. If a species or habitat has not been assessed, conducting a CCVA for that species or habitat is one of the first steps managers can take to better understand how climate will impact the focal system. Once the vulnerability assessment results have been evaluated, there are a growing number of adaptation strategies, decision frameworks and toolkits to choose from that allow managers to clarify their goals, constraints, and values that can then lead to appropriate management actions. These options for taking next steps after completing a CCVA are discussed in Chapter 4.

II. NORTHEAST REGIONAL SPECIES AND HABITATS AT GREATEST RISK AND MOST VULNERABLE TO CLIMATE CHANGE

A) OVERVIEW OF CCVA SYNTHESIS

In this section, we provide synthesized results from the 38 CCVAs that assessed RSGCN and their associated habitats in the Northeast since 2010. Species CCVAs were chosen based on if they 1) assessed an RSGCN, and 2) their geographic scope was within the Northeast or adjacent to it. Habitat CCVAs were chosen based on the same geographic criteria, including habitat classes in the Northeast associated with RSGCN (TCI & NFWDTC, 2023). Results are cumulative, including rankings reported in Staudinger et al. (2015), and new results published for Northeast RSGCN and habitats through 2023.
For fish and wildlife species, we group species into major taxonomic groups including amphibians, birds, fish (freshwater and marine), freshwater mussels, insects, marine invertebrates, other invertebrates, mammals, and reptiles. These categories match the taxonomic groups used in Chapter 2. Habitat rankings are grouped into 24 regional habitat types (TCI & NFWDTC, 2023), with the original habitat name associated with each ranking preserved for finer-scale information about the habitat in question.

In this report we do not report summary statistics aggregating vulnerability results. For example, we do not report which taxonomic groups were most often ranked as vulnerable. We do not provide these statistics because there is a concern that CCVA study inputs are different enough that aggregating results could create erroneous comparisons. For example, a CCVA of a suite of species in 2050 could show those species are less vulnerable than a CCVA of species in 2100, when climate impacts are expected to worsen; the difference therein lies not in the vulnerability of the species, but in the projected time frame.

B) FISH, WILDLIFE AND HABITAT CCVAs

Our cumulative review of species CCVAs identified 20 studies, containing 479 unique assessment ranks for 196 NE RSGCN. Seven CCVAs used the CCVI tool, which uses 5 risk categories. Two CCVAs ranked species using a binomial system as having either high or low vulnerability. The remaining 11 CCVAs used unique ranking categories with 3-5 risk levels. Thirteen studies were published since 2015 including three state-wide assessments, seven regional-scale assessments, and two global assessments. One assessment was limited to representative Army Installations across the United States (Hohmann & Wall, 2017). To note, NOAA released an interactive web tool for its published CCVAs, including several synthesized in this document (https://www.fisheries.noaa.gov/data-tools/climate-vulnerability-assessment-tool).

About half of ranked NE RSGCN have been assessed multiple times (52%; 101 species) with birds (N=104) and fishes (N=104) being ranked most often, each representing 22% of the total rankings. Twenty-one bird species and 23 fish species were ranked more than once. Almost all ranked species had at least one ranking indicating they were vulnerable or highly vulnerable to climate change (90%; 176 species). Seventy-six species (39%) had one ranking indicating they were some of the most vulnerable to climate change. About a third of all species (30%; 58
species) have had at least one ranking that indicates the species is stable or could experience a slight increase in population. A total of 221 RSGCN species (53%) have not been assessed for their climate vulnerability to date.

Table 3.3. Number and percentage of Climate Change Vulnerability Assessment rankings for RSGCN by taxon. Percentages were rounded to a tenth of a percent.

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Number of rankings</th>
<th>Percent of total rankings</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bird</td>
<td>104*</td>
<td>21.7%</td>
</tr>
<tr>
<td>Fish</td>
<td>104*</td>
<td>21.7%</td>
</tr>
<tr>
<td>Mammal</td>
<td>61*</td>
<td>12.7%</td>
</tr>
<tr>
<td>Mollusc</td>
<td>52*</td>
<td>10.9%</td>
</tr>
<tr>
<td>Reptile</td>
<td>50*</td>
<td>10.4%</td>
</tr>
<tr>
<td>Insect*</td>
<td>78*</td>
<td>16.3%</td>
</tr>
<tr>
<td>Amphibian*</td>
<td>21*</td>
<td>4.4%</td>
</tr>
<tr>
<td>Marine Invertebrate*</td>
<td>5*</td>
<td>1%</td>
</tr>
<tr>
<td>Crustacean</td>
<td>4</td>
<td>0.8%</td>
</tr>
</tbody>
</table>

*Indicates the species in the taxon were ranked more than once

While it is beyond the scope of the present synthesis to compare within-species differences, the appendices provide interactive data tables to explore particular species and taxa groups from different studies, and consider the specific factors affecting species vulnerability across different geographies, scope, and timeframes.
Our cumulative review of habitat-focused CCVAs in the NEAFWA region identified 18 studies, with 11 published since 2015. Five of the 11 recent studies ranked a suite of habitat types across state or regional scales and assigned categorical vulnerability rankings to each. The other six new CCVAs took a different approach, each assessing the vulnerability of one habitat type in distinct locations throughout its range. These CCVAs used quantitative scales of vulnerability and were not directly comparable to the other studies.

The most common method employed for Northeast habitats were the Ecosystem Vulnerability Assessment Approach (EVAA) for three regional CCVAs of forest habitats conducted by NIACS (Butler et al., 2015; Butler-Leopold et al., 2018; Janowiak et al., 2018) and the Sea Level Affecting Marshes Model (SLAMM) for tidal wetland CCVAs (Ekberg et al., 2017; Elsey-Quirk et al., 2022; Woodland et al., 2017). Other methods included the Soil and Water Assessment Tool (SWAT) for riverine habitats (Chambers et al., 2017) and the Habitat Climate Change Vulnerability Index (HCCVI) (Comer et al. 2022).

Using the classification system for habitats associated with Northeast RSGCN (TCI & NFWDTC, 2023), Forests and Woodlands (N=10 studies) and Tidal Wetlands and Flats (N=9 studies) were the most assessed habitat classes overall, occurring both in habitat CCVAs with a broad scope and more narrowly focused assessments. Subterranean Areas and Great Lakes were

**Figure 3.4. Bar plot of the number of Climate Change Vulnerability Assessment ranks of Northeast Regional Species of Greatest Conservation Need for each taxon.**
the only natural habitat classes with no CCVA results in the NE region. In addition, one anthropogenic habitat class, Plantations & Orchards, was not assessed.

C) OVERVIEW OF APPENDICES

Results from one CCVA or an array of CCVAs can help evaluate multiple species simultaneously and consider actions that may have community or ecosystem-level benefits. For example, looking at highly vulnerable species in a certain habitat or geographic area can lead managers to explore if they share any underlying causes of vulnerability.

When looking across multiple CCVAs, species might have different vulnerability rankings based on different inputs (Wade, Hand, Kovach, Muhlfeld, et al., 2017). Foden and Young (2016) provide a helpful diagram for how to interpret the results of multiple CCVAs, considering both the vulnerability ranking and the uncertainty of the rankings. These differences can stem not just from the inputs, but also from multiple vulnerability frameworks that can give different scores to the same species (Wheatley et al., 2017).

The following appendices synthesize CCVA results for Northeast RSGCN (and habitats) only. Additional (non-RSGCN species) rankings from studies prior to 2015 can be found in Appendix 2 of Staudinger et al. (2015).

i. Appendix 3.1

Appendix 3.1 provides detailed information about each of the new species and habitat CCVA studies compiled since 2015. These entries include time frame and the specific climate and non-climate factors considered. For CCVAs published prior to 2015, refer to Appendix 2.1 in Staudinger et al. (2015).

ii. Appendix 3.2

Appendix 3.2 provides guidance on how the original unique ranking methods correspond to a new condensed vulnerability ranking scale that was developed to aid in comparing results across multiple methods. Our ranking scale has five qualitative categories: Highly Vulnerable, Vulnerable, Stable, Slight Increase, Large Increase.
The number and type of ranking categories varied across the 30+ vulnerability assessments published to date, with the number of qualitative categories ranging from 2-7 per method. Due to the wide range of ranking categories and overlapping or similar category descriptions, we created a new condensed scale to facilitate comparisons of results across studies. This new condensed scale presented here builds on the methodology used in Staudinger et al. (2015) to reduce the total number of equivalent vulnerability categories from nine categories to five. This new scale provides an appropriate amount of information to quickly understand a species' or taxon’s vulnerability; however some information provided by the original CCVA may be reduced. Therefore, we provide original CCVA results in Appendix 3.4, as a reference for how the original CCVA authors ranked each species.

iii. Appendix 3.3

Appendix 3.3 is in an Excel format and has three tabs. The first tab is a table that summarizes which RSGCN are assessed in each species CCVA. The second tab is the dataset used to create the table and contains results and contextual information for all species. The third tab is a list of all Northeast RSGCN, indicating whether or not they have been ranked in a regionally relevant CCVA.

iv. Appendix 3.4

Appendix 3.4 has four tabs. The Species_ExcludingN&F2016 tab contains the data we used to create Excel pivot tables and slicers in the following tabs. Because this data set was used to create the slicers, it is formatted so rankings show up intuitively in the following tabs. The Species_ExcludingN&F2016 can be used to find all the information for each ranking.

The slicers were designed so that users can easily filter the data into more manageable subsets based on interest level. Users can select multiple categories to filter the data by using the upper right icon with the green check marks. Filters can be cleared by using the filter icon (x). Because the filters allow users to make multiple selections, it is important to note what filters are in use to ensure the displayed data is correct. For example, filtering first by a CCVA and then a subtaxon will not show all results for the subtaxon, and for those results, the CCVA filter will first need to be cleared.
The Subtaxon tab presents vulnerability rankings with subtaxon as the smallest unit of measure to look at the data. The rankings in this tab are displayed using the new condensed scale (of five categories of vulnerability), and the slicers allow for exploring subsets of information (Subtaxon and Taxon) based on the original CCVA dataset. As users make data selections, a bar chart displays the number of rankings by subtaxon. This tab is envisioned to provide an initial overview of the data; for example, a quick exploration of vulnerability rankings for salamanders.

The Species tab presents vulnerability rankings with the species (displayed by common name) as the smallest unit of measure. The slicers in the Species tab allow users to select results based on additional contextual information. For example, this tab allows for users to see vulnerability rankings for all species ranked at the southern edge of their range.

v. Appendix 3.5

One species CCVA that is excluded from the condensed ranking scale and the slicers is Nadeau and Fuller (2016). Nadeau and Fuller (2016) used a continuous scale for ranking species vulnerability, which is in contrast to most other CCVAs which use categorical ranks. Because their qualitative ranks identify which species in their list is most and least vulnerable, versus ranking species as vulnerable or not to a specific set of study conditions, we report their continuous variable ranking system separately in Appendix 3.5, and have excluded their qualitative ranks from Appendix 3.4.

vi. Appendix 3.6

Appendix 3.6 summarizes which regional habitat classes are assessed in each habitat-focused CCVA.

vii. Appendix 3.7

Habitat CCVAs that assessed a suite of habitats using a categorical scale of vulnerability are synthesized in Appendix 3.7, using the condensed vulnerability ranking scale described in Appendix 3.2. Several tabs including the full synthesized data set, slicers, and pivot tables allow users to interact with the data and sort by habitat class, geographic region, etc. Information on the remaining new habitat CCVAs (Chambers et al., 2017; Ekberg et al., 2017; Elsey-Quirk et
al., 2022; Montefiore et al., 2023; Raposa et al., 2016; Woodland et al., 2017) can be found in Appendix 3.1.

III. LITERATURE CITED


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https://doi.org/10.25923/9SB9-NJ66

https://doi.org/10.1111/gcb.15768


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APPENDIX 3.1: SPECIES AND HABITAT CLIMATE CHANGE VULNERABILITY ASSESSMENTS (CCVAS) SINCE 2015

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I. Species CCVAs

Bohm et al. (2016)

**Title:** Hot and bothered: Using trait-based approaches to assess climate change vulnerability in reptiles

**Citation:** Bohm, M., Cook, D., Ma, H., Davidson, A. D., García, A., Tapley, B., Pearce-Kelly, P., Carr, J. (2016). Hot and bothered: Using a trait-based approaches to assess climate change vulnerability in reptiles. Biological Conservation 204, 32-41.

**DOI:** [https://doi.org/10.1016/j.biocon.2016.06.002](https://doi.org/10.1016/j.biocon.2016.06.002)

**Geographic Scale:** Global

**States encompassed:** All

**Ecological Scale:** Taxa - IUCN Red List Reptiles; 1498 species included

**Projected time frame:** 1975 - 2050

**Climate models used:** Four General Circulation Models (UKMO HadCM3, MPIM ECHAM5, CSIRO MK3.5 and GFDL CM2.1)

**Climate factors assessed:** exposure to sea level rise; changes in mean temperature; temperature variability change; changes in mean precipitation; and precipitation variability change

**Non-Climate factors included:** None

**How vulnerability was defined:** “Trait-based approaches collate data concerning different ‘dimensions’ of climate change vulnerability, typically including species' sensitivity, adaptability, and exposure to climate change” (page 33).

**Measure of uncertainty:** Experts qualitatively ranked data quality as high, medium or low

**Method:** The authors conducted a trait-based global climate change vulnerability assessment. Information on traits was collected through a database on reptile traits developed for previous studies, open-access spatial data and a literature search.

**Recommendations:** Authors recommend further research to better understand traits affected by climate change, specifically dispersal ability and dependence on environmental triggers.
Brinker et al. (2018)

**Title:** Climate change vulnerability assessment of species in the Ontario Great Lakes Basin


**Link:** https://www.natureserve.org/sites/default/files/ccrr-48_1.pdf

**Geographic Scale:** The study focused on the Ontario Great Lakes Basin in Canada.

**States encompassed:** The study ends at the Province of Ontario in Canada border, but the Ontario Lake basin includes Wisconsin, Indiana, Michigan, Ohio, Pennsylvania, New York, Minnesota, Illinois

**Ecological Scale:** species-level (280 species in Ontario’s Great Lakes basin)

**Projected time frame:** 2041–2071

**Climate models used:** The study used three General Circulation Models (GCMs) - CanESM2, HadGEM2-ES, and MIROC5.

**Climate impacts assessed:** The study assessed the vulnerability of 280 species to climate-related impacts: changes in temperature and moisture

**Non-Climate factors included:** The study included non-climate factors such as land use change, invasive species, habitat degradation, and habitat fragmentation as potential stressors.

**How vulnerability was defined:** “Vulnerability is the degree to which a species is susceptible to and unable to cope with the adverse effects of climate change” (page 2).

**Measure of uncertainty:** The study quantified uncertainty by conducting sensitivity analyses. Uncertainty in available data and/or limits to current knowledge about biological traits is factored in by scoring multiple categories for a single sensitivity factor.

**Method:** NatureServe Climate Change Vulnerability Index (CCVI; Canadian v.3.0; Young and Hammerson 2015).

**Recommendations:** The study recommended that conservation efforts should prioritize the species identified as most vulnerable in the study, as well as focus on conserving and restoring habitat connectivity to help species move and adapt to changing conditions. They also suggested
improving monitoring and data collection efforts to better track species responses to climate change.

Culp et al. (2017)

Title: Full annual cycle climate change vulnerability assessment for migratory birds


DOI: https://doi.org/10.1002/ecs2.1565

States encompassed: Upper Midwest and Great Lakes (UMGL) region of the United States. (parts of Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, and Wisconsin, and three Bird Conservation Regions (boreal hardwood transition, lower Great Lakes/St. Lawrence plain, and prairie hardwood transition))

Ecological Scale: Species level (46 species of migratory birds)

Projected time frame: 2040–2069

Emissions scenario: A2 high-emissions scenario (IPCC SRES 2000)

Climate models used: Climate Wizard (Nature Conservancy’s online tool), data originated from an ensemble of 16 general circulation models downscaled by the Nature Conservancy

Climate impacts assessed: The study assessed the vulnerability of migratory birds to climate change by examining the impacts of changes in temperature and precipitation on the availability and quality of breeding and wintering habitats, timing of migration, and survival rates.

Non-Climate factors included: The study included non-climate factors such as land-use change and habitat loss that could interact with climate change to further exacerbate the vulnerability of migratory birds.

How vulnerability was defined: “We define vulnerability as the evidence that climate change or other anthropogenic factors will negatively affect a regional population of a species.”

Measure of uncertainty: Author scored the data uncertainty on a 5-point scale
Method: The study used a vulnerability assessment framework that incorporated information on species’ life histories, climate change impacts, and non-climate factors to assess the vulnerability of migratory birds to climate change. Additionally, expert consultation was used to select criteria for the vulnerability framework.

Recommendations: The study recommends that conservation efforts focus on protecting and restoring critical habitats, reducing non-climate stressors, and monitoring migratory bird populations to better understand and respond to the impacts of climate change. Specifically, because the assessed birds migrate internationally, the authors recommend managers creating international conservation efforts.

Galbraith and Morelli (2017)

Title: Vulnerabilities to Climate Change of Massachusetts Animal Species of Greatest Conservation Need.

Citation: Galbraith, H., & Morelli, T. L. (2017). Vulnerabilities to Climate Change of Massachusetts Animal Species of Greatest Conservation Need. Massachusetts Division of Fisheries & Wildlife, Natural Heritage & Endangered Species Program.

Geographic Scale: Massachusetts

States encompassed: Massachusetts

Ecological Scale: The study analyzed the potential impact of climate change on 163 animal species of greatest conservation need (SGCN) in Massachusetts

Projected time frame: 2070

Climate models used: The study used climate data from two models: the Community Earth System Model (CESM) and the Model for Interdisciplinary Research on Climate (MIROC5).

Climate impacts assessed: Changes in temperature, precipitation, sea level rise, and extreme weather events

Non-Climate factors included: The study also considered non-climate factors that may impact the vulnerability of the species, such as land use changes and habitat fragmentation.

How vulnerability was defined: “...two main components of vulnerability:
- The sensitivity of the species to both the direct effect of changing climatic factors on the organisms, and indirect effects as the changing climate impacts essential components of the species’ habitat.
- The adaptive capacity of the species (i.e., its ability to respond effectively to the changing climate by, for example, changing its behavior to lessen the impacts)” (page 4).

**Measure of uncertainty:** The experts assessing traits also scored their level of uncertainty

**Method:** The study used a vulnerability assessment framework to evaluate the potential impacts of climate change on SGCN species in Massachusetts, incorporating data from climate models, species distribution models, and expert opinions.

**Recommendations:** None

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Glick et al. (2015)

**Title:** Climate Change Vulnerability Assessment for Tennessee Wildlife and Habitats


**Link:** https://www.tn.gov/content/dam/tn/twra/documents/swap/tn_swap_vulnerability_assessment.pdf

**Geographic Scale:** Tennessee

**States encompassed:** Tennessee

**Ecological Scale:** 119 species of greatest conservation need in Tennessee

**Projected time frame:** 2050

**Climate models used:** Climate Wizard projections for 2050; IPCC Fourth Assessment Report (IPCC 2007) for a 16-model Ensemble Average

**Emissions scenario:** IPCC medium emissions scenario A1B

**Climate impacts assessed:** Change in temperature and moisture, Exposure to sea-level rise; Predicted impacts of land use changes due to human response to climate change

**Non-Climate factors included:** Distribution relative to natural barriers, distribution relative to anthropogenic barriers
How vulnerability was defined: A combination of exposure, sensitivity and adaptive capacity

Measure of uncertainty: Incorporated into the CCVI method

Method: Trait based; CCVI

Recommendations: “The results of the climate change vulnerability assessment highlighted in this report will play a crucial role in helping Tennessee shape meaningful strategies to address the additional conservation challenges posed by climate change in its ongoing efforts to protect the state’s rich biodiversity. Ultimately, addressing climate change will help improve the forecast for Tennessee’s species and habitats and ensure that the many benefits they provide for society will endure for generations to come."

Hare et al. (2016)

Title: A Vulnerability Assessment of Fish and Invertebrates to Climate Change on the Northeast U.S. Continental Shelf


DOI: https://doi.org/10.1371/journal.pone.0146756

Geographic Scale: Northeast U.S. Continental Shelf

States encompassed: Gulf of Maine to North Carolina

Ecological Scale: Species level - 82 marine fish and invertebrate species

Projected time frame: Used climate projections between 2005–2055 to evaluate climate change and decadal variability in the 20–40 year time frame through the end of the 21st century

Climate models used: The authors used 25–35 global climate models (GCMs), as well as National Marine Fisheries Service (NMFS) Climate Vulnerability Assessment Methodology (see figure 1).

Climate impacts assessed: ocean surface temperature (upper 10 m), ocean surface salinity (upper 10m), surface air temperature, precipitation, surface pH (upper 10 m), currents, and sea-level rise.
Non-Climate factors included: None

How vulnerability was defined: “We define vulnerability as a change in a species’ productivity and or abundance associated with a changing climate, including both climate change and decadal climate variability” (page 3).

Measure of uncertainty: The authors assessed uncertainty in the vulnerability assessments by considering the range of outcomes across the GCMs used in the analysis.

Method: The study used a vulnerability assessment framework that combined exposure, sensitivity, and adaptive capacity to assess the vulnerability of fish and invertebrate species to climate change.

Recommendations: The authors recommend further assessments incorporate additional climatic factors, such as ocean acidification, and have an expert panel with a broader background. They also suggest links between different assessment types, such as Habitat Vulnerability Assessments and Social Vulnerability Assessments.

Hohmann and Wall (2017)

Title: Multiscale Assessment of Listed and At-Risk Species’ Climate Change Vulnerabilities


DOI: https://doi.org/10.21079/11681/23814

Geographic Scale: Regionally representative Army Installations

States encompassed: Gulf of Maine to North Carolina

Ecological Scale: Species level -Species federally listed as threatened or endangered that are known to occur on Army Installations

Projected time frame: 2050

Climate models used: Not Stated

Climate impacts assessed: Changes in temperature and moisture, sea level rise

Non-Climate factors included: Distribution relative to barriers, anthropogenic land use change

How vulnerability was defined: A function of sensitivity, exposure and adaptive capacity
Measure of uncertainty: Unknown

Method: Trait based, CCVI. Vulnerability was calculated for both a species’ entire range and vulnerability for all Federal parcels within a species’ range

Recommendations: This approach is suitable for integrating climate change considerations into installation Integrated Natural Resource Management Plans, for evaluating whether climate change-driven impacts to listed species will affect installation resilience, and for identifying potential conservation partners among Federal land managers.

Lettrich et al. (2023)

Title: Vulnerability to Climate Change of United States Marine Mammal Stocks in the Western North Atlantic, Gulf of Mexico, and Caribbean


DOI: https://doi.org/10.1371/journal.pone.0290643

Geographic Scale: Western North Atlantic, Gulf of Mexico and Caribbean Sea

States encompassed: Atlantic coastal states from Maine to Virginia

Ecological Scale: Stock level -Cetacean and pinniped stocks managed by NOAA fisheries in the Western North Atlantic, Gulf of Mexico and Caribbean. In cases where species of the same genus were commonly grouped in the literature, the authors grouped species within the same genus into one stock.

Projected time frame: 2055

Climate models used: Coupled Model Intercomparison Project phase 5 (CMIP5), with a concentration pathway (RCP) 8.5.

Climate impacts assessed: Circulation (wind-driven, tidal), sea level rise, sea surface air temperature, temperature, precipitation, salinity, ocean acidification, sea ice cover, dissolved oxygen

Non-Climate factors included: None

How vulnerability was defined: A function of sensitivity, exposure and adaptive capacity
Measure of uncertainty: Authors conducted a bootstrap analysis to assess the certainty of the vulnerability scores by sampling with replacement the tallies for each exposure sensitivity factor. They recalculated the exposure, sensitivity and vulnerability score for each of 10,000 iterations. They compared the final scores with the bootstrapped scores, and report score certainty as the proportion of those 10,000 iterations that scored in each bin.

Method: Trait based

Recommendations: The authors recommend using the results to inform regionally specific management of these marine mammal stocks, and to repeat the assessment as better climate projections or biological information becomes available.

Maryland SWAP (2015)

Title: Maryland Department of Natural Resources

Citation: Maryland Department of Natural Resources. (2016). Maryland State Wildlife Action Plan. Annapolis, Maryland.

Link: https://dnr.maryland.gov/wildlife/Documents/SWAP/SWAP_AppendicesChapter6.pdf

Geographic Scale: Maryland

States encompassed: Maryland

Ecological Scale: Maryland SGCN, habitats, tree species

When was it conducted: 2016

Projected time frame: Not stated

Climate models used: Not stated

Climate impacts assessed: The study assessed the impacts of climate change, including changes in temperature and precipitation.

Non-Climate factors included: Not stated

How vulnerability was defined: Not stated

Measure of uncertainty: Unknown

Method: Trait based; CCVI
Nadeau and Fuller (2016)

Title: Combining landscape variables can improve utility of climate change vulnerability assessments.


DOI: https://doi.org/10.1111/gcb.13586

Geographic Scale: The study was conducted in the northeastern United States.

States encompassed: Virginia to Maine (NE AFWA)

Ecological Scale: 113 species of birds, herpetofauna, and mammals.

Projected time frame: Not stated

Climate models used: Not stated

Climate impacts assessed: The study assessed the impacts of climate change, including changes in temperature and precipitation.

Non-Climate factors included:
- Local landscape resistance
- Topoclimate homogeneity
- High elevation

How vulnerability was defined: Not stated

Measure of uncertainty: Not stated

Method: The study combined a trait-based assessment and an assessment of multiple landscape variables, including climate, topography, and land use, to assess vulnerability.

Recommendations: The study recommended that vulnerability assessments consider multiple landscape variables, rather than relying solely on climate data, to better understand and address climate change impacts on ecosystems.
Title: Global assessment of marine and freshwater recreational fish reveals mismatch in climate vulnerability and conservation effort


DOI: https://doi.org/10.1111/gcb.15768

Geographic Scale: Global

States encompassed: All

Ecological Scale: 415 recreational fish species in marine and freshwater ecosystems

Projected time frame: 2030 and 2075

Climate models used: “The study used three climate models: an ensemble of all available models for marine and freshwater climate variables under two RCPs (4.5 and 8.5) …” (page 4802).

Climate impacts assessed: The study assessed the vulnerability of recreational fish species to climate change using three climate impact categories: habitat suitability, phenology, and thermal stress.

Non-Climate factors included:
- Socioeconomic value
  - Importance for human use and
  - Cultural significance- based on the number of languages and common names associated with a species
- Conservation effort
  - Measured by tallying the number of conservation measures currently in place for each species
  - Estimating the extent of a species’ range that overlapped with a protected area

How vulnerability was defined: The study assessed three dimensions of vulnerability—sensitivity, adaptive capacity, and exposure, although did not specifically define the term.

Measure of uncertainty: Uncertainty is calculated with in the CCVI method

Method: Trait based; CCVI
**Recommendations:** The study recommends that conservation efforts for recreational fish species should consider their vulnerability to climate change impacts, and that a coordinated effort is needed to address the mismatch between vulnerability and conservation efforts.

Sneddon and Galbraith (2015)

**Title:** Climate change vulnerability assessments in the Appalachian Landscape Conservation Cooperative region

**Citation:** Sneddon, L. A., & Galbraith, H. S. (2015). Climate change vulnerability assessments in the Appalachian Landscape Conservation Cooperative region. Natural Areas Journal, 35(2), 195-207.


**Geographic Scale:** The study conducted an assessment across the Appalachian Landscape Conservation Cooperative Region.

**States encompassed:** Appalachia Georgia, Alabama, South Carolina, North Carolina, Tennessee, West Virginia, Virginia, Kentucky

**Ecological Scale:** 137 plant species and 563 animal species

**Projected time frame:** 2050

**Climate models used:** “We used a rapid assessment tool to assesses habitats, a modification of the NEAFWA (Northeast Association of Fish and Wildlife Agencies) Habitat Vulnerability model developed by the Manomet Center of Conservation Sciences” (page 3).

**Climate impacts assessed:** Temperature, moisture

**Non-Climate factors included:** Distribution relative to barriers, anthropogenic land use change

**How vulnerability was defined:** “Vulnerability was determined as a function of low to high adaptive capacity and positive to negative potential impacts” (page 27).

**Measure of uncertainty:** Assessments of vulnerability were ranked with High, Medium, or Low certainty.
Method:
- Literature review of past and current climate change research.
- Analysis of temperature and precipitation data from weather stations.
- Species distribution modeling to predict how species' ranges may shift due to climate change.
- Development of climate change vulnerability assessments for specific species and ecosystems.
- Engagement with stakeholders, such as land managers and conservation organizations, to gather information on current management practices and identify potential adaptation strategies.

Recommendations: The authors recommend using multiple climate models to better understand the range of possible impacts for one geographic location. They believe their results can help identify areas for conservation actions. They also recommended developing adaptation strategies that incorporate both ecological and social factors, such as increasing public education and outreach about the impacts of climate change and fostering collaboration between researchers, land managers, and stakeholders. Finally, they suggested investing in long-term monitoring and research to improve understanding of the impacts of climate change on the region's ecosystems.

Sneddon and Hammerson (2017)

Title: Climate change vulnerability assessments of selected species in the north Atlantic LCC region


Link: https://www.sciencebase.gov/catalog/item/5a836242e4b00f54eb32993b

Geographic Scale: The study assessed the North Atlantic Landscape Conservation Cooperative Region

States encompassed: Maine, New Hampshire, Vermont, Rhode Island, New York, Massachusetts, Connecticut, Delaware, Maryland, Pennsylvania, Virginia

Ecological Scale: 44 species chosen because they were considered species of high regional concern, representative species and foundational species.

Projected time frame: 2050
**Climate models used:** “We used the data sets recommended by the CCVI, climate predictions that represent a median of 16 major global circulation models (GCMs) and a medium emission scenario (A1B) for mid-century (2050s). For available moisture, we used the Hamon AET: PET moisture metric available from NatureServe (http://www.natureserve.org/climatechange) and derived from Climate Wizard” (page 17).

**Emissions Scenario:** Medium emission scenario (A1B) for mid-century (2050s)

**Climate impacts assessed:** Changes in temperature and moisture, sea level rise

**Non-Climate factors included:** Distribution relative to barriers, anthropogenic land use change

**How vulnerability was defined:** Not specifically defined, a function of exposure, sensitivity, and adaptive capacity

**Measure of uncertainty:** No specific measure of uncertainty, but sources of uncertainty are discussed in detail on starting on page 50

**Method:** Trait based, CCVI

**Recommendations:**
“Future conservation actions must account for climate change in addition to addressing existing stressors and threats. Our work here identified a number of species that are vulnerable to climate change, but their successful conservation will depend on taking an ecosystem approach to adaptation, as recommended by The National Fish, Wildlife and Plants Climate Adaptation Partnership (2012) and others:

- Focus conservation action on the habitats of fish, wildlife, and plant populations
- Support critical functions of ecosystems that support them
- Maintain or increase connectivity of habitat
- Reduce non-climate stressors – maintain or improve ecological integrity” (page 54). Additionally, the authors recommend using results to develop an implementation plan and recommend further monitoring and data needs.
II. Habitat CCVAs (traditional vulnerability ranking)

Butler-Leopold et al. 2018

**Title:** Mid-Atlantic Forest Ecosystem Vulnerability Assessment and Synthesis: A Report from the Mid-Atlantic Climate Change Response Framework Project


**DOI:** [https://doi.org/10.2737/NRS-GTR-181](https://doi.org/10.2737/NRS-GTR-181)

**Geographic Scale:** Mid-Atlantic

**States encompassed:** Pennsylvania, New Jersey, Delaware, eastern Maryland, and Southern New York

**Ecological Scale:** 11 forest ecosystems

**Projected time frame:** 2010-2039, 2040-2069, 2070-2099 (Tree Atlas model), 1980-2009 and 2070-2099 (LINKAGES model), and 2009-2099 (LANDIS PRO model)

**Climate factors assessed:** Temperature and precipitation projected from downscaled versions of GFDL A1FI and PCM B1- two climate scenarios, spanning a range of variation in possible future conditions

**Non-Climate factors included:** Soil, tree species traits and distribution, modifying factors including susceptibility to insect pests and fire, and human land use (harvesting, fragmentation) used as inputs for various forest impact models

**How vulnerability was defined:** Potential impacts on a forest ecosystem and the adaptive capacity of the ecosystem. (Potential impacts include both exposure and sensitivity factors of climate change vulnerability.)

**Measure of uncertainty:** Expert panelists evaluated the level of evidence and agreement supporting each of their vulnerability determinations. The evidence included observational, modeled, or theoretical information that contributed to a vulnerability assessment. The authors defined evidence as strong if there were multiple observations or models to support an assessment, and those observations or models had strong theoretical underpinnings. Panelists also evaluated the agreement of evidence based on if the evidence suggested similar outcomes.
The strength of the evidence and the agreement of evidence was then translated to an overall assessment of uncertainty based on a two-dimensional matrix. Confidence ratings are included with each vulnerability ranking. Then the panelists’ individual overall assessments were compared and discussed by the team of panelists, and the group determined the final uncertainty in one of five categories: low, low-moderate, moderate, moderate-high, and high

**Method:** Ecosystem Vulnerability Assessment Approach (Brandt et al. 2017), utilizing 26 experts who assessed potential impacts of climate change on forest ecosystems and each system’s adaptive capacity

**Recommendations:** The study does not recommend specific actions, but provides resources for adaptation planning (such as the Climate Change Response Framework).

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**Comer et al. 2022**

**Title:** Addressing Climate Change Vulnerability in the IUCN Red List of Ecosystems—Results Demonstrated for a Cross-Section of Major Vegetation-Based Ecosystem Types in the United States

**Citation:** Comer, P. J., Hak, J. C., & McIntyre, P. (2022). Addressing Climate Change Vulnerability in the IUCN Red List of Ecosystems—Results Demonstrated for a Cross-Section of Major Vegetation-Based Ecosystem Types in the United States. *Land, 11*(2), 302. https://doi.org/10.3390/land11020302

**DOI:** https://doi.org/10.3390/land11020302

**Geographic Scale:** Conterminous United States

**States encompassed:** Virginia, Pennsylvania, New York, New Jersey, Connecticut, Massachusetts, Vermont, New Hampshire, Maine

**Ecological Scale:** 33 upland habitat types (using NatureServe classification)

**Projected time frame:** 2035-2065

**Climate factors assessed:** The authors calculated two dimensions of climate change exposure: “suitability change, which quantifies departure from the historical range of spatial climate variability across the geographic range of that ecosystem type, and climate departure, which quantifies departure from the historical range of year-to-year climate variability at a given pixel location” (page 5).
**Non-Climate factors included:** IUCN Red List of Ecosystems (RLE) criteria, including reduction in distribution, restricted distribution, environmental degradation, and disruption of biotic processes

**How vulnerability was defined:** Used the Habitat Climate Change Vulnerability Index (HCCVI) definition of vulnerability, function of exposure, sensitivity and adaptive capacity

**Measure of uncertainty:** Acknowledged the qualitative nature of assessing environmental degradation in relation to a threshold of ecosystem collapse

**Method:** NatureServe’s Habitat Climate Change Vulnerability Index (HCCVI) was applied across the distribution of each habitat, and the resulting vulnerability assessments were incorporated into the RLE criterion C2 to calculate final Red List status.

**Recommendations:** The authors recommend using the vulnerability results to inform conservation planning, such as actions that increase resilience where ecosystems have low vulnerability.

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Farr et al. 2021

**Title:** An assessment of marine, estuarine, and riverine habitat vulnerability to climate change in the Northeast U.S.


**DOI:** https://doi.org/10.1371/journal.pone.0260654

**Geographic Scale:** Northeast U.S. Shelf Ecosystem, from Cape Hatteras through the Gulf of Maine

**States encompassed:** Virginia, Delaware, Maryland, New Jersey, New York, Connecticut, Rhode Island, Massachusetts, New Hampshire, Maine

**Ecological Scale:** 52 marine, estuarine and coastal riverine habitats

**Projected time frame:** 2100
Climate factors assessed: Sea surface temperature, bottom temperature, sea surface salinity, bottom salinity, pH, precipitation, air temperature, streamflow (floods and droughts), river temperature, sea level rise.

Non-Climate factors included: Anthropogenic factors: dredging/filling, pollution/eutrophication, invasive species, harmful algal blooms, shoreline hardening and synergistic effects

How vulnerability was defined: The vulnerability of a habitat is “function of the interplay between the sensitivity and the potential exposure to future change” (page 4)

Measure of uncertainty: Experts assigned each exposure and sensitivity attribute a data quality score. The authors conducted a bootstrap analysis for cases that were at the border between vulnerability categories.

Method: The authors used a hybrid approach that adapted elements from NOAA’s Fish Stock Climate Vulnerability Assessment and a habitat vulnerability model developed for the Northeastern Association of Fish and Wildlife Agencies. The hybrid approach used a trait-based framework including the sensitivity and exposure components of climate vulnerability. The authors did not have a separate metric for adaptive capacity, and instead stated it is confounded with habitat sensitivity. Sensitivity and Exposure were each ranked as Low, Moderate, High or Very High vulnerability based on a logic rule score of the component attributes. Five experts scored the sensitivity and exposure of each system.

Recommendations: The authors recommend using habitat vulnerability to more fully understand species’ vulnerability. They also believe the results can be used to update designations for Essential Fish Habitat and Habitat areas of Particular Concern.

Janowiak et al. 2018


DOI: https://doi.org/10.2737/nrs-gtr-173

Geographic Scale: New England and northern New York
**States encompassed:** Connecticut, Maine, Massachusetts, New Hampshire, northern New York, Rhode Island, and Vermont

**Ecological Scale:** 8 forest ecosystems

**Projected time frame:** 2010-2039, 2040-2069, 2070-2099 (Tree Atlas model), 1980-2009 and 2070-2099 (LINKAGES model), and 2009-2099 (LANDIS PRO model)

**Climate factors assessed:** Temperature and precipitation projected from downscaled versions of GFDL A1FI and PCM B1 - two climate scenarios, spanning a range of variation in possible future conditions

**Non-Climate factors included:** Soil, tree species traits and distribution, modifying factors including susceptibility to insect pests and fire, and human land use (harvesting, fragmentation) used as inputs for various forest impact models

**How vulnerability was defined:** Potential impacts on a forest ecosystem and the adaptive capacity of the ecosystem. (Potential impacts include both exposure and sensitivity factors of climate change vulnerability.)

**Measure of uncertainty:** Expert panelists evaluated the level of evidence and agreement supporting each of their vulnerability determinations. The evidence included observational, modeled, or theoretical information that contributed to a vulnerability assessment. The authors defined evidence as strong if there were multiple observations or models to support an assessment, and those observations or models had strong theoretical underpinnings. Panelists also evaluated the agreement of evidence based on if the evidence suggested similar outcomes. The strength of the evidence and the agreement of evidence was then translated to an overall assessment of uncertainty based on a two-dimensional matrix. Confidence ratings are included with each vulnerability ranking. Then the panelists’ individual overall assessments were compared and discussed by the team of panelists, and the group determined the final uncertainty in one of five categories: low, low-moderate, moderate, moderate-high, and high

**Method:** Ecosystem Vulnerability Assessment Approach (Brandt et al. 2017), utilizing 20 experts who assessed potential impacts of climate change on forest ecosystems and each system’s adaptive capacity

**Recommendations:** The study does not recommend specific actions but provides resources for adaptation planning (such as the Climate Change Response Framework).

Sneddon and Galbraith 2015

**Title:** Climate change vulnerability assessments in the Appalachian landscape conservation cooperative region: phase II results of assessments
Geographic Scale: Appalachian region. The authors divided the assessment into three subregions: Central Appalachian, Cumberland-Southern Appalachian, Interior Low Plateau

States encompassed: New York, Pennsylvania, Ohio, West Virginia, Virginia, Maryland, North Carolina, South Carolina, Tennessee, Kentucky, Georgia, Alabama

Ecological Scale: Habitats associated with the selected species

Projected time frame: 2050

Climate factors assessed: Temperature, moisture

Non-Climate factors included: None

How vulnerability was defined: The authors did not directly define vulnerability but said “Assessing the vulnerability of a habitat tells us its degree of resistance to change, where it exists now by examining the degree of exposure to predicted climate change, and evaluating the sensitivity of individual species comprising it” (page 41).

Measure of uncertainty: Assessments of vulnerability were ranked with High, Medium, or Low certainty.

Method: Expert panel and literature review, trait based method that scores habitats on a continuous scale.

Recommendations: The authors recommend combining the vulnerability scores with conservation status to inform conservation planning. They also recommend conducting more in-depth studies of species and habitats that were highly or extremely vulnerable.
III. Habitat CCVAs (place-based/local vulnerability ranking)

Chambers et al. 2017

**Title:** Simulating Climate Change Induced Thermal Stress in Coldwater Fish Habitat Using SWAT Model

**Citation:** Chambers, B., Pradhanang, S., & Gold, A. (2017). Simulating Climate Change Induced Thermal Stress in Coldwater Fish Habitat Using SWAT Model. *Water, 9*(10), 732. https://doi.org/10.3390/w9100732

**Geographic Scale:** Wood-Pawcatuck and Cork Brook watersheds in Rhode Island

**States encompassed:** Rhode Island

**Ecological Scale:** Freshwater watersheds with cold water habitat for brook trout

When was it conducted:

**Projected time frame:** 2010-2039; 2040-2067; 2070-2099

**Climate factors assessed:** Stream temperature, streamflow, stressful events for Rhode Island brook trout defined as days with high or low flow and temperatures exceeding 21°C.

**Non-Climate factors included:** Surface water runoff, potential evapotranspiration

**How vulnerability was defined:** Did not focus paper on vulnerability, instead on number of days brook trout experience stressful events

**Measure of uncertainty:** Incorporated into the model

**Method:** Modeled number of stressful days based on climate change projections using the Soil and Water Assessment Tool

**Recommendations:** The authors recommend further research, specifically a subbasin analysis, to identify sites that need the most protection. This could help conservation organizations that own small properties better allocate their resources.

Ekberg et al. 2017

**Title:** Development and Application of a Method to Identify Salt Marsh Vulnerability to Sea Level Rise

DOI: https://doi.org/10.1007/s12237-017-0219-0

**Geographic Scale:** Entire state of Rhode Island

**States encompassed:** Rhode Island

**Ecological Scale:** 31 salt marshes, subdivided into 39 total units

**Projected time frame:** 2100

**Climate factors assessed:** Sea level rise impacting marsh elevation and marsh loss

**Non-Climate factors included:** plant community, soil degradation/strength

**How vulnerability was defined:** The authors created an index of vulnerability by scoring the exposure variables and averaging them to create a final score. A principal component analysis (PCA) incorporating the same factors was also conducted as an alternative method to score vulnerability by interpreting the first component.

**Measure of uncertainty:** Compared measurements of elevation (GPS vs. LiDAR), and cross-validated factors included in vulnerability scores

**Method:** On-site transects to determine marsh vegetation community type, marsh soil integrity, marsh elevation. On-site data was combined with the Sea Level Affecting Marshes Model (SLAMM) in two novel vulnerability indices.

**Recommendations:** The authors recommend using the vulnerability scores to prioritize interventions for certain marshes. Which could be focusing on marshes with the highest risk, which they found were on Rhode Island’s south coast. Or focusing on marshes with low vulnerability who may be more likely to withstand impacts of climate change. The authors identified Potowomut and the Prudence Island marshes as stable, and marshes that would benefit from these actions.

The authors also recommend using the marsh assessments to identify marshes to protect across a range of habitats. For example, identifying marshes that are suitable for bird nesting or provide nekton habitat. The authors recommend this approach to maintain ecosystem functions of the marsh.

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Elsey-Quirk et al. 2022
Title: Relationships between ecosystem properties and sea-level rise vulnerability of tidal wetlands of the U.S. Mid-Atlantic


DOI: https://doi.org/10.1007/s12237-017-0219-0

Geographic Scale: Delaware Estuary and Barnegat Bay

States encompassed: New Jersey, Delaware, Pennsylvania

Ecological Scale: Coastal lagoon, coastal plain estuaries, tidal freshwater and saline marshes

Projected time frame: Current vulnerability, based on retrospective changes between 1970 and 2015

Climate factors assessed: None

Non-Climate factors included: soil bulk density, below ground plant biomass, human modification

How vulnerability was defined: Recent rates of marsh loss are used as a surrogate for vulnerability

Measure of uncertainty: Standard error was included with parameter estimates to portray uncertainty.

Method: The authors conducted field measurements to assess soil bulk density and live belowground biomass. These were used as inputs into the SLAMM model. The SLAMM model has rates of marsh loss at the dependent variable, and has a suite of marsh vulnerability indices, such as soil bulk density, to explain variation in the rates of marsh loss.

Recommendations: The authors found that differences in geomorphology lead to different vulnerability of marshes. They found that tidal freshwater marshes seem to be less vulnerable than salt marshes, and marshes with alterations to control mosquitos increased vulnerability to the impacts of sea level rise. The authors recommend restoring natural hydrology in those marshes. In other areas of high loss, such as estuaries, the authors suggest oyster restoration or living shorelines to reduce loss.
Montefiore et al. 2023

**Title:** Vulnerability of Estuarine Systems in the Contiguous United States to Water Quality Change Under Future Climate and Land-Use


**DOI:** https://doi.org/10.1029/2022EF002884

**Geographic Scale:** Contiguous U.S.

**States encompassed:** West Virginia, Virginia, Delaware, Maryland, Pennsylvania, New Jersey, New York, Connecticut, Rhode Island, Massachusetts, Vermont, New Hampshire, Maine

**Ecological Scale:** 112 U.S. estuaries and their watersheds

**Projected time frame:** modeled past (1990-2020) and future (2035-2065) periods

**Climate factors assessed:** Precipitation and marine upwellings, as they contribute to nutrient loading

**Non-Climate factors included:** Land use-land cover (LULC) change, specifically surface runoff, as it relates to nutrient loading

**How vulnerability was defined:** “Vulnerability is driven by three key dimensions: exposure (i.e., magnitude and extent of exposure to change-driven impacts), sensitivity (i.e., how responsive a system is to exposure to hazards), and adaptive capacity (i.e., potential ability and opportunities to adapt or accommodate the combined effects of the exposure and sensitivity of the system; Nelitz et al., 2013).”

**Measure of uncertainty:** Calculated 10th and 90th percentile estimates for Total Nitrogen (TN) and Total Phosphorous (TP) in addition to averages for each model, representing a range of climate impacts on future nutrient loads; acknowledged uncertainty in study scale and models and inputs used.

**Method:** This study calculated exposure by modeling LULC, precipitation, and marine upwellings in a “top-down” approach using four LULC scenarios and 20 climate scenarios. Sensitivity was based on each estuary’s present eutrophication and rate of flushing. Adaptive capacity scores incorporated both human and natural indicators in a novel approach, including “access to scientific knowledge,” “legislative/governmental actions,” and density of wetlands, with component indices each scored from 0 to 1. All three components of climate change vulnerability (exposure, sensitivity, and adaptive capacity) were incorporated into the final estimates of nutrient loading.
**Recommendations:** “Additional work should be pursued to further develop the vulnerability framework, notably by incorporating more local indicators (e.g., local policies, species tolerances to shifting seasonality in nutrients), climate projections that account for extreme precipitation events, and time-varying model parameters (e.g., EMCs). We also propose that future research build on our findings by considering how projected nutrient load changes could drive cascading ecological impacts such as vulnerability to harmful algal blooms and hypoxia.” (21)

---

Raposa et al. 2016

**Title:** Assessing tidal marsh resilience to sea-level rise at broad geographic scales with multi-metric indices


**DOI:** https://doi.org/10.1016/j.biocon.2016.10.015

**Geographic Scale:** 16 National Estuarine Research Reserves across the U.S.

**States encompassed:** Washington, Oregon, California, Alaska, Texas, Wisconsin, Ohio, Maine, New Hampshire, New York, Massachusetts, Rhode Island, New Jersey, Maryland, Virginia, North Carolina, South Caroline, Georgia, Alabama, Louisiana, Florida.

**Ecological Scale:** Tidal Marsh

**Projected time frame:** Used past data to calculate marsh elevation change over time, but did not project into the future

**Climate factors assessed:** Sea level rise

**Non-Climate factors included:** marsh elevation, marsh elevation change over time, accretion and sediments, tidal range

**How vulnerability was defined:** A function of sensitivity and exposure. Sensitivity can include proximity to riverine sediment sources, tidal range, anthropogenic alterations. They use indices of sensitivity and exposure to create indices for resilience which they define “the ability of a system to resist and recover from perturbation,” and understand as “the inverse” of vulnerability (page 264).
**Measure of uncertainty:** Authors acknowledge uncertainty around future SLR rates, but do not quantify it; summary statistics include error bars representing standard error.

**Method:** The authors scored individual metrics and assigned three score definitions by looking at the range of variation in the data across 16 marshes. MARS risk index was based on the concept that a low score for any of the five categories represents a risk in the face of SLR and multiple low scores represent a higher risk. They summed the number of categories, and created a MARS average index, and calculated the MARS ratio index by dividing the rate of marsh elevation change by the long-term rate of SLR.

**Recommendations:** The authors suggested different management actions for three tiers of MARS indices. For marshes that have consistently high MARS indices they suggested preservation efforts. For marshes with moderate scores they suggest efforts to enhance resilience to sea level rise. Then, for marshes with low MARS indices scores, or marshes that are unlikely to exist at the end of the century due to sea level rise, they suggest management efforts focus on transformation, such as upland marsh migration.

---

Woodland et al. 2017

**Title:** Changes in Habitat Availability for Multiple Life Stages of Diamondback Terrapins (*Malaclemys terrapin*) in Chesapeake Bay in Response to Sea Level Rise

**Citation:** Woodland, R. J., Rowe, C. L., & Henry, P. F. P. (2017). Changes in Habitat Availability for Multiple Life Stages of Diamondback Terrapins (*Malaclemys terrapin*) in Chesapeake Bay in Response to Sea Level Rise. *Estuaries and Coasts, 40*(5), 1502–1515. https://doi.org/10.1007/s12237-017-0209-2

**DOI:** https://doi.org/10.1007/s12237-017-0209-2

**Geographic Scale:** Chesapeake Bay

**States encompassed:** Maryland

**Ecological Scale:** Observed nest site of the Diamondback Terrapin in the Chesapeake Bay and the multiple coastal habitat types those sites are in: estuarine beach, salt marsh, brackish marsh and transitional salt marsh and aggregate undeveloped dry land (a category which includes several habitat types including forestland, grassland, wetlands, crop/pastureland, partially developed land, and shrubland)

**Projected time frame:** 2100

**Climate factors assessed:** Sea Level Rise
Non-Climate factors included: None

How vulnerability was defined: Did not define

Measure of uncertainty: The authors identified uncertainty inherent to data inputs in the discussion section, but did not list a specific measure of uncertainty.

Method: Predicted Sea Level Rise was determined by Sea Level Affecting Marshes Model (SLAMM), then combined with data from a field survey of diamondback terrapin nesting.

Recommendations: The authors recommended further research into the life history of diamondback terrapin to better understand how the population will be effected by sea level rise. They suggest because there are uncertainties in the life-history, conservation planning should take a conservative approach. For example, the authors suggest minimizing development on the coast and areas directly inward of coastal wetlands to allow for habitat to migrate landward as sea level rises. They note more information on life-history is needed before designing more active conservation approaches, such as beach nourishment.
CHAPTER 4: CLIMATE CHANGE ADAPTATION STRATEGIES AND RESOURCES IN THE NORTHEAST UNITED STATES

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†Current address

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Key findings

- Over the last 10 years, the field of climate adaptation, the intentional, planned adjustments to natural and human systems to reduce the risks from climate change, has advanced from establishing theory to implementation.

- We present 9 climate adaptation decision support tools that can help clarify objectives, evaluate trade-offs to identify no-regrets decisions and prioritize actions along with links to additional case studies of how the tools have been implemented on the ground.

- We share 17 new Climate Adaptation Action Menus that provide brainstorming prompts for climate-informed management approaches that can be integrated into a range of planning processes. We provide links to access the menus, as well as details on menu topic and how they could be integrated into adaptation planning.

- We present 7 summaries of emerging climate adaptation topics to offer ways to meet goals developed by using the decision support tools and add more detail to some of the tactics presented.

- We propose a process for developing climate-informed actions for SWAP reports.

- We present an example approach of how to apply the 5Ws approach, which is a set of reflective questions to consider different dimensions of a climate plan, to develop actions.

- We provide three case studies illustrating actions that can simultaneously address both climate and non-climate threats, based on a survey of taxa team experts.

- In the appendix, we detail 81 additional climate adaptation resources, including Tribal Adaptation Plans, and 14 case studies of how decision support tools were used by researchers and managers.

I. THE CURRENT LANDSCAPE OF CLIMATE ADAPTATION

A) INTRODUCTION TO CLIMATE ADAPTATION

i. Overview of the chapter

Over the last 10 years, one of the biggest advances in the field of climate adaptation has been the progression from theory to implementation. In the 2015, State Wildlife Action Plans (SWAP), many decision tools and frameworks were released, alongside a limited set of case studies of implementation for fish, wildlife, and their habitats (Staudinger et al., 2015). Today,
there are a more examples of on-the-ground implementation of adaptation tools and new decision support resources stemming from an improved understanding of the impacts of climate change (LeDee et al., 2021; Thompson et al., 2021). While these resources represent a new frontier in adaptation planning, they build on the foundation of adaptation principles that remain relevant today (Stein et al. 2014). These various resources can be used alone or in combination to meet national climate adaptation goals (NFWPCAP, 2012) and tailored to local contexts. We outline and discuss these resources in the sections of this chapter and provide case studies of their implementation in Appendix 4 to help state managers incorporate climate change into the 2025 SWAP revisions.

In Section I, we detail resources that are designed to assist managers in understanding tradeoffs, addressing uncertainty, and developing future-oriented plans. Decision support tools can help identify big-picture goals and select on-the-ground actions, while adaptation menus offer lists of specific climate-informed strategies and approaches. The resources highlighted in Section I are either relatively new, being used in new ways, or have become more prominent since the regional climate synthesis provided for the last SWAP revision cycle (Staudinger et al. 2015). We include new tools and topics that provide managers novel management approaches to explore and apply to their system of interest. We conclude this section with a discussion on emerging adaptation topics.

In Section II, we demonstrate two possible ways to integrate climate adaptation frameworks and tools into the SWAP planning process. We outline a process using the “5Ws” as a set of reflective questions (who, what, where, when and why), as described by Oakes et al. (2022), to integrate climate adaptation tools into the development of species- and habitat-specific, climate-informed actions. Then, we demonstrate how combining information on "threat multipliers" of climate change (e.g., increased pollution from changing precipitation regimes) can help identify actions that address a broader suite of species and conservation challenges. We believe this approach could help state agencies address interacting and indirect threats of climate change. This chapter is accompanied by an appendix of resources and Northeast-specific case studies to further demonstrate their applicability to Regional Species of Greatest Conservation Need (RSGCN) and associated habitats.
ii. Climate adaptation background

Handler et al. (2022) defines climate adaptation as “the intentional, planned adjustments to natural and human systems to reduce the risks from climate change.” When adopting an adaptation approach, managers are often confronted with addressing uncertainties, making trade-offs, and managing for multiple objectives (Reside et al., 2018). While historically, wildlife conservation has been species focused (e.g., Endangered Species Act), due to the impact of climate change some researchers advocate for focusing management on sustaining geophysical properties that foster biodiversity rather than maintaining for specific species composition (Taylor, 2016 Stein et al., 2013). Focusing on geophysical properties or ecosystem functions may become more pertinent as species and populations move across political boundaries or go locally extinct (Stein et al. 2013). Determining the focus of climate-informed adaptation may involve nuanced discussions that consider trade-offs when developing goals, identifying management priorities, and grappling with the social and political dimensions of climate adaptation (Hemming et al., 2022).

Adding to the complexity of these trade-offs, managers are tasked with developing goals and strategies in the face of profound abiotic, biotic, and social uncertainties (Reside et al., 2018). For example, we cannot precisely predict the future levels of global greenhouse gas emissions change because policies and societal responses are constantly in flux, nor do we know how biotic systems will cope through novel responses (Reside et al., 2018). Beyond the direct impacts of changing temperature and precipitation regimes there are innumerable indirect effects of climate change, such as predicted increases in human wildlife conflict (Abrahams et al. 2023), increased runoff and nutrient loading in coastal systems (Montefiore et al., 2023), and spread of non-native invasive species (Schweiger et al., 2010). Therefore, decision support tools that are robust to a range of possible futures, identify tradeoffs, and clearly link goals to actions, can guide managers towards intentional decisions that can be routinely revisited and adjusted as new information becomes available (Hemming et al., 2022). Frameworks such as Resist-Accept-Direct or Resistance-Resilience-Transformation are a few of the resources presented in this chapter that provide a process to move through complex and uncertain management challenges. A one-size-fits-all approach may not be realistic and the social and political values of the surrounding community are important considerations when developing and implementing effective climate adaptation strategies (Orlove et al., 2020; Webster et al., 2017).
B) DECISION SUPPORT TOOLS

Climate-informed decision support tools can help managers by providing ways to categorize actions or desired ecosystem states. The following frameworks help meet Element 4 of the SWAP requirements by setting big-picture goals, evaluate trade-offs to identify no-regrets decisions and prioritize actions (Table 4.1).

Table 4.1. Outline of the climate adaptation decision support tools discussed in this chapter with information on the primary topic areas each tool was developed to address and how to access the resources. Resources are ordered newest to oldest based on year published or application to Northeast natural resources.

<table>
<thead>
<tr>
<th>Application information</th>
<th>Tool or Framework</th>
<th>Year first published or adapted to conservation</th>
<th>Focal topic area</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tools published since 2015 that have been implemented in the Northeast</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Witness, Acknowledge, Mend, Protect, Unite, Move (WAMPUM)</td>
<td>2021</td>
<td>Tribal adaptation to sea level rise</td>
<td></td>
<td>Leonard, 2021</td>
</tr>
<tr>
<td>Resist, Accept, Direct (RAD)</td>
<td>2020</td>
<td>Categorizing future oriented management actions</td>
<td></td>
<td>Thompson et al., 2020</td>
</tr>
<tr>
<td>Conservation Standards</td>
<td>2020</td>
<td>Guidance for all steps in adaptation planning</td>
<td></td>
<td>GIZ &amp; CMP, 2020</td>
</tr>
<tr>
<td>Tools presented in Staudinger et al. (2015) with new applications</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Resistance, Resilience, Transformation (RRT)</td>
<td>2007</td>
<td>Desired ecosystem states</td>
<td></td>
<td>Millar et al., 2007</td>
</tr>
<tr>
<td>Portfolio Planning</td>
<td>2004</td>
<td>Biotic, abiotic, and genetic diversity</td>
<td></td>
<td>Figge, 2004; Dufour et al., 2015</td>
</tr>
<tr>
<td>Method</td>
<td>Year</td>
<td>Description</td>
<td>Reference</td>
<td></td>
</tr>
<tr>
<td>---------------------------------------------</td>
<td>------</td>
<td>--------------------------------------------------</td>
<td>------------------------------------</td>
<td></td>
</tr>
<tr>
<td>Scenario Planning</td>
<td>2003</td>
<td>Developing future scenarios for creative planning</td>
<td>Peterson et al., 2003</td>
<td></td>
</tr>
<tr>
<td>Structured Decision Making (SDM)</td>
<td>1986</td>
<td>Reproducible and transparent decision-making processes</td>
<td>Walters 1986; Runge 2011</td>
<td></td>
</tr>
<tr>
<td>New tools not yet used in the Northeast</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5Ws</td>
<td>2019</td>
<td>Assessing management actions</td>
<td>Oakes et al., 2021</td>
<td></td>
</tr>
<tr>
<td>Nature Futures Framework</td>
<td>2020</td>
<td>Societal values of nature and future states</td>
<td>Pereira et al. 2020</td>
<td></td>
</tr>
</tbody>
</table>

i. **Witness, Acknowledge, Mend, Protect, Unite, Move (WAMPUM)**

The WAMPUM framework incorporates an Indigenous worldview into coastal climate adaptation to address the unique political context and challenges of Atlantic Coast Tribal nations (Leonard, 2021). The framework, developed by Kelsey Leonard, a water scientist, legal scholar, and enrolled citizen of the Shinnecock Indian Nation, supports adaptation to the impacts of sea-level rise (SLR) for Tribal nations located along the Atlantic Coast within a broader framework of Indigenous water and climate justice.

The components of the WAMPUM framework are:

- **Witness**: Witness warnings from human and non-human relations and follow the patterns of the natural world.
- **Acknowledge**: Acknowledge traditional teaching and restore cultural stewardship practices.
- **Mend**: Mend the shoreline and practice healing for coastal and environmental sovereignty.
- **Protect**: Protect future generations by protecting cultural sites and ancestors.
- **Unite**: Unite with other communities to build capacity, plan relocation.
- **Move**: Move to new places with cultural connections and rebuild.
This framework can help state agencies reflect and assess if they are undertaking adaptation actions that could be implemented in collaboration with Indigenous communities. For example, the WAMPUM framework details how non-human relations play an important role as beings that can communicate warnings, such as unusual whale mortality events (Leonard, 2021). In this example, state agencies could partner with Tribal nations to provide data, collect data, or support Tribal nations in collecting their own data to witness these warnings. Additionally, this framework can help state agencies reflect and assess if they are undertaking adaptation actions that might unintentionally harm Indigenous communities. For example, developing a voluntary buyout system that scatters a community rather than relocating it as a whole may disrupt cultural and social connections (Leonard, 2021; Marino, 2018). While this framework does not replace formal collaboration with Tribal nations, it can offer a supplemental pathway for guidance and reflection.

ii. Resist-Accept-Direct (RAD)

The Resist-Accept-Direct (RAD) framework consists of three general management pathways that can be used when responding to climate change. The three pathways were designed to be all-encompassing (e.g. all decisions could fall under Resist, Accept, or Direct) (Lynch et al., 2021; Schuurman et al., 2022; Thompson et al., 2021). While RAD options are mutually exclusive, they can be used in a single landscape on different spatial or temporal scales; for instance, resisting change in the short-term with plans to direct change in the long-term. Additionally, these climate-informed approaches can be used in concert with other decision support tools to incorporate climate trajectories in traditional decision support structures, such as adaptive management (Powell et al., 2017; Schuurman et al., 2022; Lynch et al. 2022). The RAD framework is a unique tool because the focus is on management actions rather than the goal state.
for an ecosystem (Schuurman et al., 2022).

**Figure 4.1.** Conceptual diagram depicting the three pathways for the Resist, Accept and Direct (RAD) framework. Retrieved from: https://www.usgs.gov/media/images/rad-conceptual-figure.jpg

The RAD framework was designed to help managers move beyond traditional conservation actions, such as managing for historical baselines, to also consider future conditions in setting action targets (Lynch et al., 2021). The RAD framework encourages managers to consider not just the conventional approach of resisting change, but also the pathways of accepting and directing change. RAD reframes accept as an active decision, rather than a passive one, potentially reducing the taboo of taking no action (Thompson et al., 2021). While climate change is bringing new challenges to managers, researchers have found that, to date, conservation professionals are generally implementing conventional conservation actions (LeDee et al., 2021; Lynch et al., 2022; McLaughlin et al., 2022; Reside et al., 2018).

The RAD approach built from the Resistance-Resilience-Transition typology (see below) to emphasize what managers can control—their actions—rather than what they cannot—i.e., outcomes under changing climate. In practice, RAD is a tool for discussing and categorizing management options and encourages managers to identify their own desired end-states. It is notable, that RAD centers on the actions of a natural resource manager, and this represents
particular perspective of engaging with the natural world that may not translate to other viewpoints. For example, many Indigenous people see themselves as in relationship with non-human beings, not as external forces developing actions to intervene in the natural world (Shultz et al., 2022). One potential way to identify if RAD is the right framework for a system could be by reflecting on the system and stakeholders central to adaptation planning process. Clifford et al. (2021) discusses how external influences, such as faith, risk perception or focal species of concern, can shape a manager's decision-making process. Reflecting on these influences could help managers make transparent decisions and consider how others with differing worldviews may react to different actions.

Table 4.2. A list of recent examples of implementing the RAD framework in the Northeast or adjacent regions for natural-resource management.

<table>
<thead>
<tr>
<th>Topic</th>
<th>Case study</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Invasive species in marine systems</td>
<td>Lionfish in the Caribbean</td>
<td>Rogers-Bennett et al., 2022</td>
</tr>
<tr>
<td>Invasive species in freshwater systems</td>
<td>Introduced Brook Trout vs. Native Bull Trout</td>
<td>Dunham et al., 2022</td>
</tr>
<tr>
<td>Invasive species in a Tribal context</td>
<td>Ojibwe knowledge holder’s perspectives on terrestrial invasive species</td>
<td>Shultz et al., 2022</td>
</tr>
<tr>
<td>Agricultural systems and private landowners</td>
<td>Categorizing coastal farmers and woodlot managers in response to the potential for rapid and irreversible change to the landscape due to SLR in Maryland and Virginia</td>
<td>Sudol et al., 2023</td>
</tr>
<tr>
<td>River systems</td>
<td>Understanding management actions along the different areas of the Mississippi River Basin</td>
<td>Ward et al., 2023</td>
</tr>
</tbody>
</table>
### iii. Resistance Resilience Transformation (RRT)

The RRT framework is a well-established tool to help managers define the highest level of desired future conditions for climate-informed management where resistance, resilience, or transition are three options. As described in Swanston et al. (2016), these three options are:

- To enhance an ecosystem’s ability to **resist** change
- To enhance an ecosystem’s ability to be **resilient** and accommodate some degree of change
- To intentionally anticipate or accommodate change and actively facilitate change and **transition** within the ecosystem

Most managers are trained to manage resources toward desired ecological conditions, and the RRT framework allows for a climate-focused lens through which managers develop their plans, such as silviculture prescription writing (for more examples see Evans et al. 2021). Additionally, the RRT pathways allows managers to define their goals internally, and clearly communicate these goals externally. Much like the RAD framework, the RRT pathways can be used in combination with different options implemented at different spatial and temporal scales (Millar et al., 2007).

The RRT framework was developed to help managers move away from paradigms that rely on historical conditions and instead embrace a flexible approach that “emphasizes ecological processes rather than structure” (Millar et al., 2007, p. 2146). Millar et al. (2007) conceptualized the framework to help develop forest management strategies for managing forested ecosystems in the context of climate change and the associated uncertainties. The RRT framework has since...
been adapted by several organizations, such as the Northern Institute of Applied Climate Science (NIACS) and the U.S. Forest Service, to incorporate the resources they created into decision-making processes (Swanston et al., 2016).

A limitation of the RRT framework is that the “resilience,” as an adjective, is subject to individual interpretation, which can be loaded with cultural connotations (Fisichelli et al., 2016; Maltby et al., 2023; Meerow & Newell, 2019). A resilient system may not always be a desired end point. For example, a resilient socio-ecological system could perpetuate long-standing inequities (Meerow & Newell, 2019). Meerow and Newell (2019) recommend asking resilience for whom, what, when, where and how as a reflective exercise to create an equitable and clear definition of resilience.

Table 4.3. A list of recent examples of implementing the RRT framework in the Northeast for natural-resource management.

<table>
<thead>
<tr>
<th>Topic</th>
<th>Case study</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forestry</td>
<td>Restoring Red Spruce Forests</td>
<td>The Nature Conservancy</td>
</tr>
<tr>
<td>Forestry</td>
<td>Adaptive Silviculture for Climate Change</td>
<td>Palik et al., 2022</td>
</tr>
<tr>
<td>Forestry</td>
<td>Preserving ash in the face of the emerald ash borer and climate change</td>
<td>D’Amato et al., 2023a, 2023b</td>
</tr>
<tr>
<td>Marsh management</td>
<td>Using the Adaptation Workbook to create a resilient marsh and maintain habitat for a suite of birds</td>
<td>Allouez Bay Marsh Bird Habitat Restoration Planning</td>
</tr>
</tbody>
</table>

**iv. Portfolio planning**

Fundamental to portfolio planning is risk diversification, the idea that managers should spread exposure to risk so that declines or losses of a single conservation “asset” will not cascade to the entire focal area or resource. This idea was adapted from Modern Portfolio Theory (MPT) used in finance (Markowitz 1991), to optimize conservation activity by strategically allocating
conservation resources to maximize the expected conservation returns for a given level of uncertainty (Ando & Mallory 2012). Due to uncertainty in species’ and ecosystems’ responses to climate change, managers may choose to maintain a portfolio of genetic, biotic, or geophysical diversity, for example by working to restore the genetic diversity of a population, by maintaining a diverse plant community, or by protecting a landscape with varied topography (Anderson et al., 2015; Webster et al., 2017). Managers could additionally implement a portfolio of different management actions over a landscape scale to reduce risk (Aplet & Mckinley, 2017).

Additionally, portfolio planning can integrate into adaptive management by trying different things in different places and learning over time (Allen et al., 2013; Aplet & Mckinley, 2017).

There are multiple ways to approach portfolio planning. At the species level, managers can assess and preserve a portfolio of assets that a species can exploit throughout time and space rather than focusing on preserving optimal resources (Schindler et al., 2015). At a population or community level, managers can create a portfolio plan to ensure species are distributed across a gradient of climatic or geophysical conditions (Webster et al., 2017). Another is to assess the genetic diversity of different populations, and how they contribute to the adaptive potential of the species as a whole. Trout Unlimited took this approach for a portfolio plan of the eastern range of Brook Trout (Salvelinus fontinalis; Fesenmyer et al., 2017). They assessed the genetic diversity of different population patches of Brook Trout and the habitat condition of those patches (Fesenmyer et al., 2017). This allowed them to prioritize stream restoration to maintain a portfolio of genetic diversity (Fesenmyer et al., 2017).

In the Northeast, portfolio planning has been useful for anadromous River Herring (Alosa spp.) populations. River Herring spawn in coastal pond systems that vary widely in size and environmental conditions across the Northeast U.S. Migration timing into these pond systems is highly variable and has shifted in some but not all monitored systems (Dalton et al., 2022; Ellis & Vokoun, 2009; Huntington et al., 2003; Lombardo et al., 2020). Adult migration timing has shown both advancements and delays likely due to the interaction of climate change and barriers to aquatic connectivity (e.g., dams and culverts) (Legett et al., 2023). High phenological variation can be beneficial if it corresponds to a species’ flexibility in response to different environmental conditions (i.e., phenotypic plasticity). However, high variability can create challenges for making reliable predictions. In this scenario, portfolio planning could be useful to identify multiple systems across River Herring’s spawning range that support diverse thermal
habitat conditions and span the full breadth of their spawning phenological window. A coastwide conservation plan could also factor in temporal buffers that account for increasing phenological variation and the occurrence of extreme events (e.g., spring Nor'easters and other storms) to proactively protect the species from additional stressors that disrupt or delay migration (e.g., water withdrawals and coastal development projects) during this critical life history period.

v. Climate change scenario planning

Scenario planning is a collaborative exercise that allows participants to explore various futures, integrate climate and non-climate threats, and develop strategies to achieve a goal for the focal system or species (Butler et al., 2020). In this process, managers consider the different ways climate change could manifest and explore the effects of multiple potential future conditions on important resources. They select climate models whose estimated future climate conditions (i.e., climate futures) are likely to result in scenarios that are:

- **Plausible**, given what we know about climate and atmospheric science
- ** Relevant** to resources the users care about
- **Divergent**, so the models give different results from one another (e.g., one model that predicts reduced spring rainfall and moderate increase in extreme storm frequency and another with wetter springs and stronger increase in storm frequency and strength)
- **Challenging** to entrenched assumptions about the future and best management strategies.

It is an appropriate approach for complex systems with drivers of change that are highly uncertain and highly impactful, and particularly useful if managers have limited or no control over external factors (Symstad et al., 2017). Numerous recent applications of scenario planning are available including for endangered species (Borgaard et al. 2019, 2022) and protected areas (Star et al., 2016).

Detailed descriptions of scenario planning and various applications can be found in Staudinger et al. (2015), the AFWA Voluntary Guidance (2023), and by the National Park Service (Star et al., 2016). Briefly, scenario planning was originally developed for military planning and has more recently been applied to natural-resource management in a climate context (Walker et al., 2003; Duinker & Greig, 2007). It is helpful for developing “no-regrets” decisions that are robust to different future conditions. Participants can think creatively and reduce uncertainties by categorizing drivers of change (e.g. climate change, social change) along two axes into generalized storylines that represent potential futures of climate and non-climate
threats (Star et al., 2016). Scenario planning allows managers to explore different storylines that include a variety of climate projections (Symstad et al., 2017). For example, if precipitation projections for an area of interest have a high range of uncertainty that span both ends of extreme precipitation - drought and flooding, scenario planning creates a structure to think about managing for either extreme. Another axis is developed for a different variable with diverging outcomes, such as land protection versus land development. These two axes create four diverging scenarios (Wet and Developed, Dry and Developed, Wet and Protected, Dry and Protected) that managers can consider when developing future plans. Alternatively, participants can approach scenarios by mapping out futures they want and work backwards to understand the conditions needed to reach those futures (Star et al., 2016).

![Figure 4.2. Matrix describing four future population scenarios for North Atlantic Right Whales developed through a planning initiative led by NOAA in 2018. The yellow star in the Thrive scenario (upper right quadrant), denotes the scenario considered by workshop participants as the best possible future for Right Whales. Figure modified from Borggaard et al. 2020.](image)

An important consideration when undertaking scenario planning is that it is a social exercise, and, as such, is influenced by the participants and their affiliations. Butler et al. (2020) reported that scenario planning workshops often do not immediately reach the decision-making
stage because social dynamics of participants influenced outcomes. One way to navigate the social aspects of scenario planning is to employ a skilled facilitator (Butler et al., 2020).

Successful workshops of scenario planning have been led by NOAA for two endangered species (North Atlantic Right Whale (*Eubalaena glacialis*) and Atlantic Salmon (*Salmo salar*), resulting in the identification of high priority research needs and management targets (Borggaard et al., 2019, 2022). The outcomes of the Atlantic Salmon scenario planning workshop resulted in climate-related actions being integrated into the species’ Recovery Plan for the first time (USFWS and NMFS 2019). Both scenario planning workshops led to increased collaboration and coordination with external partners and the outputs from both workshops were used to develop climate change priorities for other planning, monitoring, and management documents. For example, the outputs from North Atlantic Right Whale scenario planning exercise led to increased coordination with Canadian partners, funding dedicated to understanding prey changes, and the initiation of a *North Atlantic Right Whale Emergency Response Plan* (Borggaard et al., 2021).

**Table 4.4.** A list of recent examples of implementing scenario planning in the Northeast for natural-resource management.

<table>
<thead>
<tr>
<th>Topic</th>
<th>Case study</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cultural Resource</td>
<td>Acadia National Park</td>
<td>Star et al., 2016</td>
</tr>
<tr>
<td>Terrestrial Mammal</td>
<td>Moose in New York State</td>
<td>Connelly et al., 2020</td>
</tr>
<tr>
<td>Marine Mammal</td>
<td>Atlantic Right Whale</td>
<td>Borggaard et al., 2021</td>
</tr>
<tr>
<td>Diadromous Fish</td>
<td>Atlantic Salmon</td>
<td>Borggaard et al., 2019</td>
</tr>
</tbody>
</table>
vi. Structured Decision Making (SDM)

The U.S. Geological Survey and the U.S. Fish and Wildlife Service have worked together to adapt areas of decision science for natural resource management. The result, Structured Decision Making (SDM), is based on the PrOACT (Problem, Objectives, Alternatives, Consequences, Tradeoffs) model; it is a transparent, collaborative process for navigating complex management problems involving multiple stakeholders and/or shared governance. It can incorporate expert elicitation or sophisticated quantitative performance models to predict, evaluate, and compare consequences across a suite of objectives and alternatives (e.g., multi-criteria decision analysis tools). Although not designed specifically for the context of climate change, the issues that arise in the context of climate adaptation, including multiple objectives, uncertainty, risk, and complex predictive models, are readily incorporated into SDM.

The benefits of SDM are that the thorough and well-articulated process results in a transparent output that can clearly show how a decision was made, and thus reduces conflict as well as some biases. Documentation of the process facilitates transparency and allows decisions made at each step to be revisited and reconsidered as needed as desirable or undesirable.

**Figure 4.3.** The steps and cycle for conducting the Structured Decision-Making process. Figure modified from Runge (2011).
outcomes unfold. Drawbacks of SDM are that it can be time and knowledge intensive. However, trainings or an outside partner can help reduce those barriers. For a further discussion of SDM see Chapter 4 of the Climate Change Synthesis Report (TCI & NEFWDTC, 2023a).

There is an introductory online course: Structured Decision Making – Vermont Cooperative Fish and Wildlife Research Unit Spreadsheet and R Project (uvm.edu) and more information through the National Conservation and Training Center https://trainingcenter.fws.gov/courses/programs/decision-analysis/.

**Table 4.5.** A list of recent examples of implementing the SDM in the Northeast or adjacent regions for natural-resource management.

<table>
<thead>
<tr>
<th>Topic</th>
<th>Case study</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coastal species harvesting</td>
<td>Horseshoe Crab harvesting in Delaware Bay</td>
<td>McGowan et al. 2020</td>
</tr>
<tr>
<td>Coastal wetland management</td>
<td>Delaware wetlands and coastal infrastructure</td>
<td>Lyons et al. 2020</td>
</tr>
<tr>
<td>High-elevation species protection</td>
<td>Looking for optimal management options for the Shenandoah Salamander</td>
<td>Grant et al. 2014</td>
</tr>
</tbody>
</table>

**vii. Conservation Standards**

The Conservation Standards workbook was created by a collaboration between the German development agency Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) and the Climate Guidance Working Group of the Conservation Measures Partnership (CMP) (GIZ & CMP, 2020). It guides managers through the adaptation project planning process using a series of small, detailed steps. For instance, the workbook details how to develop a theory of change (defines long-term goals and then maps backwards to identify necessary preconditions), how to develop a work plan, and how to create a culture of learning within an agency. To help managers complete each step there are additional resources on the Conservation Standards website (https://conservationstandards.org/library-item/climate-smart-conservation-practice/), links to project management software, and illustrative case studies. In one case study, exercises from the guide were used by the Atlantic Coast Joint Venture to identify an initial suite of
priorities and prioritize threats to focus efforts for a salt marsh bird plan for the entire East Coast of the U.S. (Atlantic Coast Joint Venture, 2019). The workbook is extremely detailed potentially making it useful for organizations new to adaptation planning but may not be flexible enough for organizations who already have a standard planning process. However, managers can still refer to sections of Conservation Standards for insight on discrete parts of the adaptation project planning process.

viii. What, when, where, why, and who (5Ws)

The 5Ws framework enables users to rapidly assess how—and by what means—climate change will require innovation beyond business-as-usual conservation practice (Oakes et al., 2021). The assessment encourages users to ask "what, when, where, why, and who"—or the "5Ws"—as a tool in project design and implementation. The "what," for example, means considering whether climate variability and projected changes will require taking new actions or modifying existing actions. The "who" asks users to consider: by whom, with whom, who benefits and who might bear potential harm or tradeoffs from project implementation and anticipated outcomes. Oakes et al. (2021) argue that using the 5Ws can help design conservation projects, especially using Nature-based Solutions (NbS) to climate change, account for climate futures, engage communities, and deliver a comprehensive suite of benefits. This rapid assessment could also be used for near-term or small-scale planning, while flagging issues to tackle with more in-depth planning processes (Oakes et al., 2021). Other researchers have proposed the 5Ws as a method for co-producing context-specific definitions of resilience when engaging with communities and resilience planning for urban areas (Meerow & Newell, 2019) and fisheries (Maltby et al., 2023). These authors used the 5Ws to develop plans, while Oakes et al. (2021) used the 5Ws to assess actions.

For example, when planning to replace culverts, asking the 5Ws could help identify where the project is needed based on shifting precipitation, when it needs to happen to accommodate projected increased runoff, and who should be consulted (e.g. transportation departments, residents, wildlife experts). The 5Ws can also illuminate dimensions of the project that are already climate-informed. The what, replacing culverts, and the why, to upgrade failing infrastructure, do not need to be reconsidered. We suggest a process for how to apply the 5Ws to writing actions for the SWAPs reports in Section II of this chapter.
ix. Nature Futures Framework (NFF)

The Nature Futures Framework (NFF) is designed to capture three overarching viewpoints for how individuals and communities value nature: “Nature for Nature,” “Nature for Society,” and “Nature as Culture” (Pereira et al., 2020). The NFF was developed through a co-production process facilitated by the Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services. The process involved an expert group with diverse participants including Indigenous peoples, government officials, and academics (Pereira et al., 2020). The NFF helps users visualize a complex social reality where communities will likely hold some proportion of all three of these values. The NFF can help managers navigate where different communities sit on the value spectrum to identify areas of conflict or collaboration. By mapping different values, managers can use the NFF to incorporate multiple differing worldviews of nature, an often overlooked dimension of conservation planning, into structured decision-making and modeling (Palacios-Abrantes et al., 2022; Pereira et al., 2020).

Palacios-Abrantes et al. (2022) give several examples of how the NFF can help illustrate different values, and the acceptable management actions stemming from those values. Many of their examples are international, so we provide a hypothetical adaptation of one of their examples for the Northeast. This example is intended to demonstrate how the NFF could be used to map values in different communities and does not incorporate all dimensions of managing Black Ash (Fraxinus nigra).

Fraxinus nigra, commonly known as Black Ash, Brown Ash, Basket Ash, and Swamp Ash (hereafter referred to as Black Ash) is an economically, ecologically, and culturally important species in the Northeast and is currently threatened by Emerald Ash Borer (EAB; Agrilus planipennis). Climate models indicate Black Ash habitat may decrease in the future. Additionally Black Ash is intolerant of drought making it sensitive to changing hydraulic conditions (D’Amato et al. 2020). As climate change potentially reduces suitable habitat for Black Ash, management decisions may become more fraught, and creating climate-informed management plans may need to incorporate actions that align across different value systems.

- **Nature for Nature**: Black Ash is a keystone species in forested wetlands given its ability to survive in the flooded or seasonally wet conditions (Constanza, 2017). It provides habitat and forage for wildlife and plays an
important role in nutrient cycling and hydrologic regulation (D’Amato et al., 2023b).

- **Nature for Society**: Black Ash can be commercially harvested, and the presence of EAB reduces the economic prospects of Black Ash.

- **Nature as Culture**: Black Ash is an important cultural species for Indigenous peoples across its range, including the Mohawk Council of Akwesasne and the Wabanaki in the Northeast. Materials from ash are used for making baskets and snowshoes, and for medicines. Black Ash is central to creation stories for Indigenous people in the Northeast.

  In this example, the NFF framework could help managers identify that people who identified with any of the three value sets, value maintaining the tree species. It can also help identify points of divergence between the values sets. For instance, there could be disagreement about whether harvesting trees is permissible. Individuals holding Nature for Society values may support a “pre-salvage” harvesting strategy to minimize economic loss. Those holding this value may feel that there is nothing to be done to eliminate EAB, and the best course is to minimize economic losses (D’Amato et al., 2023a). In potential opposition, those with Nature for Nature and Nature as Culture values might advocate for management strategies to preserve Black Ash to maintain the ecosystem benefits and cultural practices. While aligned on maintaining Black Ash populations, those holding these different value sets might choose different ways to approach pest management. Those holding Nature as Culture values might prioritize protecting basket trees, additionally some management options, like injecting ash with insecticides, may be a difficult decision that needs community consultation (D’Amato et al. 2023b). Researchers suggest coordinating with different communities to help create an effective and integrated path forward for managing Black Ash (D’Amato et al. 2023b).

C) ADAPTATION MENUS

Adaptation menus are comprehensive, hierarchical lists of potential actions within a particular management topic or discipline (e.g., forest management, outdoor recreation). The menus provide brainstorming prompts for climate-informed management approaches that can be integrated into a range of planning processes such as the NIACS Adaptation Workbook.
Adaptation menus originated from researchers at NIACS (Swanston et al. 2016), and in the past 10 years, there has been a proliferation of climate-informed adaptation menus that provide context-specific tactics. These options can be applied locally, stepping down from broader, landscape-scale strategies that align with regional or national goals (e.g., from the NFWPCAP, 2012). Approaches from these menus can help fulfill all required elements of SWAPs.

In this section we describe and provide links to the different menus (Table 4.6 and Table 4.7). Due to the sheer number of menus and tactics we do not list tactics from the menus, and instead provide links where managers can interface with the menu, tactics, and examples directly. For more details on the menus see Appendix 4.1.

i. Menus that fit within the Adaptation Workbook format

There are currently 12 menus that have been published by NIACS and their partners. All of which follow a consistent structure of organizing adaptation actions into strategies, approaches, and tactics. Menus were designed to supplement the Adaptation Workbook, planning process (Swanston & Janowiak, 2012; Swanston et al., 2016).

Users can access NIACS menus through their web page or links in Table 4.7. The menus have:

- Detailed descriptions of strategies or tactics
- Background references and descriptions of scope (e.g. forest carbon, terrestrial wildlife)
- Hypothetical examples illustrating how the element could be applied on-the-ground
- Case studies of how items have been applied (note that not all menus have this)

The NIACS menus are published in peer-reviewed publications (Table 4.6), following testing with conservation professionals and substantial literature review. These publications give detailed background on the creation of the menu, additional references, and some publications have in-depth case studies.

The Adaptation Workbook is a flexible tool that allows a manager to incorporate their project’s context and their own judgment to move through generating goals, tradeoffs, and actions. While the flexibility allows a manager to fit the workbook to their needs, it also means users need significant knowledge about their site and management goals to move through the Workbook. Users need to begin the Workbook with this background knowledge because final
decisions on adaptation actions rely on a user’s context-specific professional judgment rather than broad recommendations (Schmitt et al., 2021).
**Table 4.6.** A list of NIACS adaptation menus, focal areas for strategies and information on how to access and interact with these tools using web links. Resources are ordered alphabetically.

<table>
<thead>
<tr>
<th>Menu</th>
<th>Scope</th>
<th>Publication</th>
<th>Year</th>
<th>Author Partner Organizations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>Covers: crops, livestock, farm enterprises</td>
<td>USDA Climate Hubs technical bulletin 1944</td>
<td>2016</td>
<td>USDA Climate Hubs, U.S. Forest Service, Northern Institute of Applied Climate Science, Agriculture Research Service, Natural Resources Conservation Service</td>
</tr>
<tr>
<td>CA forest and timberland management for state’s diverse forest ecosystems</td>
<td>USDA California Climate Hub Technical Report CACH-2020-1</td>
<td>2020</td>
<td></td>
<td>USDA Climate Hubs, U.S. Forest Service, Northern Institute of Applied Climate Science, The Nature Conservancy, John Muir Institute of the Environment, SW CASC</td>
</tr>
<tr>
<td>Fire-Adapted Ecosystems</td>
<td>Addresses land managers seeking to support ecosystem adaptation to changing fire regimes</td>
<td>Journal of Climate</td>
<td>2022</td>
<td>USDA Climate Hubs, U.S. Forest Service, Northern Institute of Applied Climate Science, Southwest FireCLiME, Northern Arizona University, University of Central Arkansas, Forest Stewards Guild, USGS, UFWS</td>
</tr>
<tr>
<td>Topic</td>
<td>Description</td>
<td>Source</td>
<td>Date</td>
<td>Collaborators</td>
</tr>
<tr>
<td>-------------------------------</td>
<td>----------------------------------------------------------------------------------------------------------------------------------------------</td>
<td>------------------------------------------------------------------------------------------------</td>
<td>--------</td>
<td>------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Forested Watersheds</td>
<td>Support hydrologic function, ecological restoration, and infrastructure improvements</td>
<td>Climate Services</td>
<td>2019</td>
<td>USDA Northern Forests Climate Hub, U.S. Forest Service, Northern Institute of Applied Climate Science, Trout Unlimited-New England, and Minnesota DNR-Forestry</td>
</tr>
<tr>
<td>Great Lakes Coastal Ecosystems</td>
<td>Great Lakes coastal ecosystems, helping coastal properties adapt to climate changes</td>
<td>USDA Northern Forests Climate Hub white paper</td>
<td>2022</td>
<td>USDA Northern Forests Climate Hub, U.S. Forest Service, Northern Institute of Applied Climate Science, U.S. FWS, Central Michigan University, U.S. EPA, Northland College, Wisconsin DNR, Environment and Climate Change Canada</td>
</tr>
<tr>
<td>Inland Glacial Lakes Fisheries</td>
<td>Glacial lakes fisheries in the upper Midwest.</td>
<td>Lake and Reservoir Management</td>
<td>2019</td>
<td>USDA Northern Forests Climate Hub, Northern Institute of Applied Climate Science, Midwest Glacial Lakes Partnership, University of Missouri, USGS, WI DNR, MN DNR, University of MN,</td>
</tr>
<tr>
<td>Non-Forested Wetlands</td>
<td>Projects that support hydrologic function, vegetation management, ecological restoration, and infrastructure improvements in non-forested wetlands.</td>
<td>USDA Northern Forests Climate Hub Report</td>
<td>2019</td>
<td>USDA Northern Forests Climate Hub, Northern Institute of Applied Climate Science, WI DNR, Northland College, GLIFWC, Vilas County (WI), Stantec</td>
</tr>
<tr>
<td>Recreation</td>
<td>Outdoor recreation planning, including recreation infrastructure.</td>
<td>Sustainability</td>
<td>2019</td>
<td>USDA Northern Forests Climate Hub, U.S. Forest Service, Northern Institute of Applied Climate Science</td>
</tr>
<tr>
<td>Tribal Perspectives</td>
<td>Incorporating Tribal perspectives into climate adaptation planning for natural and cultural resources. Also includes Guiding</td>
<td>Great Lakes Indian Fish and Wildlife Commission</td>
<td>2019</td>
<td>USDA Northern Forests Climate Hub, U.S. Forest Service, Northern Institute of Applied Climate Science, GLIFWC, 1854 Treaty Authority, Dynamite Hill Farms, Inter-Tribal Council of Michigan, College of Menominee Nation, Red Cliff Band of Lake Superior</td>
</tr>
<tr>
<td>Section</td>
<td>Description</td>
<td>Source</td>
<td>Year</td>
<td></td>
</tr>
<tr>
<td>-------------------------------</td>
<td>------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
<td>--------------------------------------------------------------------------------------------</td>
<td>------</td>
<td></td>
</tr>
<tr>
<td>Principles for working with</td>
<td>Tribal nations and Tribal citizens.</td>
<td>Chippewa, Michigan Tech University, Lac du Flambeau Band of Lake Superior Chippewa</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tribal citizens.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban Forests</td>
<td>Actions to promote human health, climate adaptation, and carbon mitigation in urban forests.</td>
<td>USDA Northern Forests Climate Hub, U.S. Forest Service, Northern Institute of Applied Climate Science, American Forests, University of Washington, Morton Arboretum, Field Museum of Natural History, University of Connecticut, NRCS,</td>
<td>2021</td>
<td></td>
</tr>
<tr>
<td>Wildlife Management</td>
<td>Terrestrial wildlife management, including population management and habitat management. Does not include practices specific to fish and aquatic wildlife.</td>
<td>USDA Northern Forests Climate Hub, U.S. Forest Service, Northern Institute of Applied Climate Science, MI DNR, MW CASC, University of Wisconsin-Madison, WICCI</td>
<td>2022</td>
<td></td>
</tr>
</tbody>
</table>
To support managers in translating these menus into management actions in a local context, NIACS provides in-person trainings, trainings and webinars on their YouTube page, and a spatially explicit map of case study examples. This includes a specific wildlife management training course, which was last offered in 2022 and archived videos can be found on the YouTube page. Additionally, NIACS sends out a listserv that gives updates and announcements on these trainings and resources (link to sign up can be found on NIACS' homepage: https://www.niacs.org)

ii. Other action-based resources

We highlight four resources that are formatted differently than the menus above, and do not use strategies, approaches, and tactics to organize their action (Table 4.7). These menus provide different levels of background information and have different organizational structures, which we describe briefly below (see also Table 4.7).

The Climate Adaptation Guidebook for Fisheries Management provides detailed background information on how climate change impacts fisheries, management opportunities and challenges, and case studies. The Climate Adaptation Toolkit for Marine and Coastal Protected Areas provides links to vulnerability planning and hosts a searchable database for adaptation actions. The Connectivity and Climate Change Toolkit provides background on adaptation planning generally and connectivity in different contexts (e.g. riparian, grassland, coastal). Additionally, it provides context-specific actions to enhance connectivity with links to additional resources. The Adaptive Capacity Management Actions Menu can be downloaded from the supporting information for the associated journal article (Thurman et al. 2022). It provides actions to meet conservation objectives that are informed by adaptative capacity, enhance adaptive capacity, with associated examples of actions, and insights on when a focus on exposure or sensitivity is more practical.
**Table 4.7.** A list of climate change adaptation menus and toolkits, focal areas for strategies and information on how to access and interact with these tools using web links. Resources are ordered by year of publication.

<table>
<thead>
<tr>
<th>Publisher</th>
<th>Name</th>
<th>Year Published</th>
<th>Topic</th>
<th>Link</th>
</tr>
</thead>
<tbody>
<tr>
<td>EcoAdapt</td>
<td>Climate Adaptation Guidebook for Fisheries Management</td>
<td>2016</td>
<td>Fisheries</td>
<td><a href="https://www.cakex.org/guide/climate-adaptation-guidebook-fisheries-management">https://www.cakex.org/guide/climate-adaptation-guidebook-fisheries-management</a></td>
</tr>
<tr>
<td>EcoAdapt</td>
<td>Climate Adaptation Toolkit for Marine and Coastal Protected Areas</td>
<td>2021</td>
<td>Marine and Coastal Protected Areas</td>
<td><a href="https://www.cakex.org/MPAToolkit">https://www.cakex.org/MPAToolkit</a></td>
</tr>
<tr>
<td>Association of Fish and Wildlife Agencies</td>
<td>Connectivity and Climate Change Toolkit</td>
<td>2021</td>
<td>Connectivity</td>
<td><a href="https://www.fishwildlife.org/application/files/9216/1582/0864/Connectivity_and_Climate_Change_Toolkit_FINAL.pdf">https://www.fishwildlife.org/application/files/9216/1582/0864/Connectivity_and_Climate_Change_Toolkit_FINAL.pdf</a></td>
</tr>
</tbody>
</table>

**D) EMERGING ADAPTATION TOPICS**

In this section we highlight a subset of emerging adaptation topics that have become prominent in the literature and the adaptation community. These topics offer a new lens on time-tested conservation techniques or represent new frontiers in management designed specifically to address the novel conditions that arise from climate change impacts. These topics can offer ways to meet goals developed by using the decision support tools above and add more detail to some of the tactics presented in the adaptation menus. The list of topics is not comprehensive but rather were assembled based on feedback and discussions over the last 10 years with state and federal conservation practitioners in the NE CASC region. Topics covered include:

- Assisted migration or conservation translocation which offers ideas for moving at-risk individuals or populations to more climate suitable areas.
- Climate change refugia conservation which offers a new way to identify what land to protect and buy time for species of conservation concern.
● Enhancing the adaptive capacity of species (or populations), including through novel interventions such as genetic engineering and provenancing.

● Tools to connect refugia or foster adaptive capacity through gene flow on a landscape.

● Nature-based Solutions as an innovative way to use natural systems as infrastructure to provide ecosystem services.

● Ways to integrate human dimensions, including specific resources for Tribal engagement, to better involve and account for human communities in climate change adaptation decision-making and outcomes.

● Suggestions for monitoring in the context of climate change.

● Recent innovations in climate adaptation (Stein et al. 2024).

i. Assisted migration/ conservation translocation

Assisted migration, which includes conservation translocation, is the intentional movement, and in some cases, relocation of a focal species to a recipient community (Cole et al., 2022). Assisted migration can be used to meet several different conservation objectives. For example, one reason for assisted migration is to move an at-risk species or population to an area that is currently—or projected to be—most climatically suitable. Another objective may be to move a species to fill ecological, cultural, economic, or other functions threatened by the loss of current species (Cole et al., 2022). Assisted migration can be carried out by moving individuals sourced from different climatic regions projected to match the new locations they are being moved to (referred to as climate-adjusted provenancing; Prober et al., 2015). The expectation of this type of assisted migration is these species will compensate for declines in current populations as climate change and other stressors advance (Meek et al., 2023), and offer a head-start for populations in responding to climate change. In cases where migrated individuals are of the same species as those they are replacing, there is an expectation that these immigrant individuals may have “pre-adapted alleles” that could increase the declining population’s adaptive capacity (Meek et al., 2023). Another way to carry out assisted migration is to move individuals from a different species into a community to fill the ecological function of a locally extinct species. In both cases, assisted migration can dovetail into portfolio planning management strategies.
Many natural-resource agencies have a long history of introducing species, often for recreational or economic purposes (Palik et al., 2022), as well as for conservation (Meek et al., 2023). For example, foresters have moved seeds from southern tree populations to northern locations to increase forest production (Palik et al., 2022). More recently, the central driver of introductions has shifted towards additional considerations of maintaining species diversity or ecosystem function under new climatic conditions (Palik et al. 2022; Cole et al., 2022). Assisted migration and conservation introduction could help managers realize goals that align with broader strategies of the RRT framework by increasing *resilience* for populations or systems. Assisted migration could also help managers *direct* change by establishing populations outside of their historical range where climatic conditions are predicted to be suitable.

However, one concern regarding assisted migration is the possibility of an introduction leading to negative unintended consequences, such as wildlife disease outbreaks (Thurman et al. *in press*). Indeed, there is a lack of long-term outcomes for climate-assisted migration experiments and a bias towards plants (Twardek et al. 2023). To minimize the likelihood of unintended consequences, managers can implement risk assessments or use climate projections in conjunction with consideration of other ecological factors of the species to model (or simulate) the outcome of conservation translocations (McLaughlin et al., 2022). In forested landscapes, modeling assisted migration strategies under multiple climate projections determined that non-endemic species did not thrive under any climate scenario and aggressive assisted migration scenarios greatly diminished species that provided important ecosystem services (Gustafson et al., 2023). The National Park Service has created a guide to help managers assess the ecological risk for a proposed assisted migration action (Karasov-Olson et al. 2021).

Recently, the Fish and Wildlife Service revised a rule on relocating endangered species (federalregister.gov/d/2023-13672). The revision allows establishing experimental populations of endangered species outside of their historic range, acknowledging abiotic conditions are changing and historic ranges may no longer be suitable. This revised rule may allow for more experimental populations and more data on what makes climate-informed assisted migration successful or unsuccessful.

**ii. Genetic interventions**

Species face myriad stressors, all of which can be exacerbated by climate change. These layered selection pressures (i.e., evolutionary forces) are rapidly exceeding the tolerance ranges
of many species (Wiens, 2016). Nevertheless, some species exhibit rapid responses to stressors via evolutionary adaptation that demonstrate their adaptive capacity, while others fail to respond. The ability of populations to evolve genetically in response to environmental change is largely determined by the amount of variation in inherent traits. Most management decisions that incorporate considerations of species’ evolutionary potential (e.g., captive breeding) have been implemented in response to imminent threats of extinction (Thompson et al., 2023). In contrast, efforts utilizing a forward-looking strategy, for example to increase or facilitate evolutionary potential (e.g., selection for species and genotypes with wide moisture and temperature tolerances); (Thurman et al., 2022) are relatively uncommon to date (Cook et al., 2021).

Genetic interventions can be accomplished through traditional and new conservation tools. Established conservation techniques, such as increasing connectivity, can help connect populations (e.g., through networks and corridors of climate refugia) and facilitate gene flow between them (McLaughlin et al., 2022). New molecular tools can help target more specific management actions, for example, via mapping genetic structure and connectivity (Thurman et al., 2022) or using gene editing to create seed material to restore ecosystems (Meek et al., 2023). DeLuca et al. (2021) found that connectivity varies greatly from species to species, and mapping and modeling tools may be needed to identify the best regional approaches to connecting refugia and facilitating gene flow. New technologies, like eDNA, are also available to help managers understand shifts in the spatial and temporal distribution of species (and genomes) and can reduce the costs of monitoring genetic diversity (Meek et al., 2023; Thompson et al., 2023).

Enhancing evolutionary potential can occur through direct interventions that include genetic rescue, captive breeding and release, or more novel approaches like genetic engineering and hybridization (Thurman et al., 2022). These interventions are often considered a last resort when a species faces extinction, because they are resource intensive and carry a high risk of unintended consequences. Certain genetic interventions can help populations adapt to a changing climate by, for example, maintaining a diversity of alleles for natural selection to act upon (McLaughlin et al., 2022; Meek et al., 2023). While conservation institutions have altered genetic diversity of populations throughout history (e.g. fish hatcheries and stocking), new technologies offer a pathway for conservation agencies to use genetic interventions to help species adapt to climate change.
While genetic interventions are a promising conservation pathway, they can be controversial and have important ethical considerations. For example, the rapid extinction rate of Hawai‘i’s endemic birds, primarily due to avian malaria (LaPointe et al., 2012), is likely to be exacerbated by climate change as range expansion of mosquitoes is forecasted to encroach into higher elevations previously considered to be disease refugia (Liao et al., 2017). Traditional approaches to controlling avian malaria, such as chemical and biological controls, have unacceptable impacts that result in mortality of non-target species. Therefore, researchers are evaluating gene-editing (i.e., genetic engineering) for malaria resistance in Hawaiian Honeycreepers (Samuel et al., 2020) as a more transformative solution to improving species evolutionary potential. However, gene-editing remains controversial, and the ethical implications need to be scrutinized alongside the eco-evolutionary and logistical considerations, which will require extensive cross-disciplinary investigation (Thurman et al. in press).

iii. Refugia

![Climate Change Refugia](https://www.usgs.gov/media/images/climate-change-refugia)

**Figure 4.4. Definition and illustration of climate change refugia (Slyngstad, B. 2023). Retrieved from: https://www.usgs.gov/media/images/climate-change-refugia.**

Climate change refugia include areas that are relatively buffered from contemporary climate change over time and enable persistence of valued physical, ecological, and socio-cultural resources (Morelli et al., 2016, 2020). Identifying refugia enables limited staff time and funding to be directed towards areas that are more likely to be valuable in terms of resource
objectives in the future. The steps include defining the objectives; assessing resource-specific climate vulnerabilities; mapping areas that are buffered from this climate exposure; and prioritizing these areas for adaptation actions. Although climate change refugia conservation will not be a blanket solution in the long-term, protecting these areas will preserve valuable adaptive capacity in the near-term and provide habitat for vulnerable species that are range shifting from other regions as they track their climate niches (Morelli et al., 2020). Protecting refugia is an option that aligns with the RAD framework by allowing managers to resist change while planning for other management pathways or options that they may choose to accept or direct. Examples of how refugia have been identified and managed around the world, and especially in the Northeastern U.S., can be found at climaterefugia.org.

Figure 4.5. The Climate Change Refugia Conservation cycle and steps. Adapted from Morelli et al. (2016).

iv. Regional connectivity for climate adaptation

For many species, predicted refugia in the Northeast may be relatively limited by mid to end of century. As a result, it will be imperative to prioritize locations on the landscape that will facilitate ecological flow over time to ensure that generational movements and metapopulation dynamics are preserved as species distributions shift to adapt to climate change. Preserving
regional connectivity is a means of facilitating natural movement of individuals, populations, and genes to allow genetic adaptation and range shifts in response to changing climate. Connectivity also serves to connect refugia not only to each other, but also ensures that locations that may offer climate refugia are accessible, potentially increasing their viability.

Prioritizing regional connectivity is thus a natural complement to assisted migration, genetic interventions, and refugia. The Designing Sustainable Landscapes project’s ecoConnect model produces maps of regional connectivity across the Northeast for suites of ecosystems that can be used to prioritize conservation and management efforts related to preserving or enhancing connectivity (https://umassdsl.org/data/ecoconnect/). The ecoConnect tool maps connectivity through the landscape as it exists now - independent from any particular conservation targets or conserved areas. The ecoConnect tool recognizes that organisms in different ecosystems have different connectivity needs and are therefore more likely to move through the landscape via preferred habitat. For example, wetland species of conservation concern are more likely to prefer and travel through wetlands during dispersal movements. Furthermore, the taxa and processes that benefit from connectivity vary tremendously in scale from highly localized short movements of slow-moving species to multigenerational gene flow in vagile species that operate over much larger distances. Therefore, ecoConnect assesses and integrates local and regional scale connectivity into a single map that is applicable to a wide range of natural process scales. Preserving connectivity hedges against the uncertainty of predicting the location and effectiveness of targeting refugia or assisting migration. Protecting regional connectivity does not require a precise understanding of how species or ecosystems will react to climate change, rather it facilitates species-determined adaptation.

Rewilding could offer a lens to shape connectivity or restoration projects. Specifically, protecting and connecting refugia could be informed by rewilding's goal of promoting self-sustaining ecosystem processes and the autonomy of biotic and abiotic landscape features (Pior & Ward, 2016). Rewilding is a form of ecological restoration aimed at restoring biodiversity and restoring natural processes, while aspiring to reduce human influence on ecosystems. It emphasizes recovering specific ecological interactions, such as the restoration of top predators, and other functions that would have maintained ecosystems prior to human influence (Carroll & Noss, 2021; Van Meerbeek et al., 2019). Researchers argue rewilding can help facilitate climate mitigation by driving carbon capture in ecosystems (Schmitz & Sylvén, 2023; Schmitz et al.)
(Carroll & Noss, 2021). However, others see the term as too imbued with cultural meanings of wilderness to be actionable (Jørgensen, 2015). For example, the concept of wilderness in North America often overlooks the history of Indigenous peoples, and how they shaped landscapes, and further promotes the divide between human and nature (Jørgensen, 2015). While rewilding could be a philosophy of conservation that synthesizes many goals and actions, if implemented in the densely populated Northeast, these projects may still need to balance human interests.

v. Nature-based Solutions (NbS)

![Figure 4.6. Graphic defining Nature-based Solutions and examples of applications across the United States (Slyngstad, B. 2023). Retrieved from: https://www.usgs.gov/media/images/nature-based-solutions.](image)

Nature-based solutions are actions and infrastructure that utilizes natural features and processes to protect, conserve, or restore ecosystems while generating a range of social benefits through ecosystem services including providing habitat for fish and wildlife, regulatory, and protective services and can increase the overall visual aesthetics of a place (Warnell et al. 2023). NbS can include actions like building blue or green infrastructure, also referred to as natural
infrastructure, that mimics natural systems. For instance, a built oyster reef can serve as a breakwater for storm surge and provides commercial aquaculture (Powell et al., 2019). It can also include traditional conservation and protection strategies informed by climate modeling. For example, Ericson (2017) modeled the impacts of different land covers and riparian buffers on future flooding conditions of Otter Creek, VT. Models estimated that by 2099 the 100-year flood will have 48% more flood flow than today’s 100-year flood, but that reforestation of the creek’s reach would decrease downstream peak flow of a 100-year flood by 14-29% (Ericson, 2017). Suggesting even small-scale implementation of NbS, in this case stream reforestation, can greatly mitigate climate-induced flooding impacts (Ericson, 2017). NbS can provide small-scale, site-specific benefits, for example, as a green roof or rain garden that cools or drains the surrounding area and provides pollinator forage; or at ecosystem-scales, for example, by restoring a floodplain to improve water quality (Pathak et al., 2022; Wilkening et al., 2022).

Integrating human dimensions, for example, using the NFF to identify community values, and environmental justice is central to implementing NbS (Warnell et al., 2023).

Powell et al. (2019) lists and links a wide range of ecological and human community benefits to natural infrastructure and NbS in coastal habitats, many of which can also be realized in terrestrial systems. Terrestrial carbon storage and blue carbon are two prominent examples of NbS benefits that have received much attention in the media and scientific literature; however, implementing these solutions may come with tradeoffs, and because of both their prominence and nuance we provide additional discussion on their applications.

a. Carbon storage vs. habitat

Balancing NbS with sustaining other important ecosystem functions and values is critical. For example, there has been an expanding focus on the development of climate mitigation strategies that enhance and maintain levels of carbon storage in a diversity of ecosystems. In terrestrial ecosystems, these strategies often emphasize management practices that increase the size of belowground and aboveground carbon stocks, such as through reducing levels of soil disturbance in agricultural systems or minimizing harvest severity and frequency in forests (Ryan et al., 2010). Forest habitats have become the primary focus of state and federal level policies for mitigating climate change given their outsized importance in terrestrial carbon storage. Correspondingly, several states and regions have proposed the cessation of harvesting on public forest lands to maximize forest carbon stores. However, there may be unintended consequences
to this approach, including reductions in suitable habitat for wildlife species reliant on young forest habitats conditions (Littlefield & D’Amato, 2022) and leakage of harvesting impacts to other regions or ownerships, such as increased harvesting in areas without restrictions to offset losses (Berlik et al., 2002). Recommendations to minimize these consequences have included continued allowance for management for a mosaic of different forest and woodland habitats on public lands, including young forest habitats, that although storing less carbon, provide important habitat conditions for several sensitive species (Littlefield & D’Amato, 2022).

b. Blue Carbon Ecosystems (BCEs)

Biomass and sediment in coastal vegetated ecosystems provide significant carbon sinks, which are commonly called Blue Carbon Ecosystems (BCEs) (Novak et al., 2020; Colarusso et al., 2023; McLeod et al., 2011). In the Northeast, BCEs are predominantly represented by seagrass meadows, most often eelgrass, and salt marshes (Colarusso et al. 2023). BCEs can sequester carbon at a higher rate than forests (McLeod et al., 2011). Large areas of BCEs have already been lost, primarily due to development, making it critical to preserve existing meadows and marshes and to restore meadows for additional carbon storage (McLeod et al. 2011, Novak et al. 2020). Protecting seagrass meadows is important because if disturbed, carbon stored in the sediment is released (Johannessen & Christian, 2023; Novak et al., 2020). For instance, (Warnell et al., 2022) modeled the coastal zone in the mid-Atlantic and showed it could switch from a carbon sink to a carbon source over the next 100 years due to disturbance. Beyond storing carbon, these habitats support biodiversity, fishing resources, and can reduce coastal erosion by absorbing wave energy and stabilizing sediment (Colarusso et al., 2023).

While BCEs are important for mitigating and adapting to climate change, they are sensitive to direct and indirect climate-related threats, meaning conservation efforts need to explicitly link to climate change impacts and projections. The projected increase in both extreme storms and heat events could damage these habitats (Colarusso et al., 2023; Novak et al., 2020; Johannessen & Christian, 2023). Eelgrass meadows are threatened by eutrophication and sedimentation, which could become greater threats as climate change is likely to increase runoff events (Novak et al., 2020). Salt marshes are also at risk due to inundation from sea level rise (Colarusso et al., 2023; Powell et al., 2017). To collect baseline data for these habitats and inform future conservation efforts in the Northeast, BCEs have been mapped in the Northeast (Colarusso et al. 2023) and Mid-Atlantic. Furthermore, Northeast Ocean Data created a website
to collect maps of eelgrass meadows; however, regular surveys are key to ensure eelgrass maps remain up-to-date due to the dynamic nature of their distribution.

Like terrestrial carbon sinks, managing BCEs to balance carbon storage and other benefits can be complex. These ecosystems are highly productive, and closures of these areas for ecological protection could disrupt local economies. BCEs are often ecological commons, where property rights are murky and the upstream actions of many affect the ecosystem, making management challenging (Merk et al., 2022). Merk et al. (2022) suggest that a way forward to managing these ecosystems is to understand local property rights where they occur, account for all the ecosystem services they provide, and design local governance structures to ensure BCEs are managed for social and ecological outcomes. Using the WAMPUM framework or NFF to reflect on potential management options along the Atlantic Coast could be another way to begin to address the social dimensions and tradeoffs of managing BCEs.

vi. Integrating human dimensions in climate adaptation planning

![Figure 4.7. Figure with vignettes illustrating interactions between people and the natural world. Figure credit Winn, C (2024). Retrieved from: https://www.usgs.gov/media/images/environmental-and-climate-justice](https://www.usgs.gov/media/images/environmental-and-climate-justice)
A core tenet of wildlife management in the United States is that wildlife are managed for the benefit of the public, and wildlife management plans and actions should incorporate public values, interests and preferences (Jacobson et al., 2022; Manfredo et al., 2021; Pomeranz et al., 2021). As climate changes, communities may feel ecological grief due to the loss of species (Cunsolo & Ellis, 2018), overwhelmed and anxious because of climate change (Brulle & Norgaard, 2019), and come into increased conflict with wildlife as species respond to climate impacts (Abrahams et al., 2023). These novel social dynamics and values will be important to capture in future management plans and actions because managers may implement novel interventions that require public support (LeDee et al., 2021; Twardek et al., 2023). Tools listed below and in Appendix 4.1 can help fulfill SWAP Elements 7 and 8.

Identifying social values and interfacing with the public is a crucial part of the job of a natural-resource manager (Bennett et al., 2022) and can improve outcomes for wildlife (Serota et al., 2023). There is an added layer of social and political complexity when managers have to, for example, navigate Tribal sovereignty or relationships between state governments and Tribal nations (Norton-Smith et al., 2016). To meet these challenges, we highlight tools in the appendices that can increase understanding of the public’s current values (such as America’s Wildlife Values; Yale Program on Climate Change Communication; The Nature of Americans), improve engagement with Tribal nations (such as Climate Change and Indigenous Peoples: A Synthesis of Current Impacts and Experiences), and improve general science communication skills (such as Compass). Additional resources can be found in Chapter 8 of the Northeast Regional Conservation Synthesis (TCI & NEFWDTC, 2023a).

vii. Monitoring

Monitoring is a critical component to SWAPs (Element 5) and encompasses methods for detecting and tracking biotic, abiotic, and social changes due to climate change, as well as evaluating outcomes of management actions (Hemming et al., 2022; Reside et al., 2018). Monitoring adaptation actions is crucial because of the potential for novel conditions, or because managers are implementing novel tactics, or conditions are changing, and the effectiveness of traditional actions is decreasing. Monitoring is an integral step in adaptive management because it allows for loop learning and adjusting actions based on feedback from management outcomes. Here we discuss some climate-specific considerations for monitoring. Additional resources that address monitoring more generally can be found in Chapter 8, section 8.5 of the Northeast
Regional Conservation Synthesis as well as a compiled list of citizen science monitoring projects and initiatives (TCI & NEFWDTC, 2023a).

Monitoring the success of adaptation actions is tied to project objectives and goals. For example, a project implemented to increase connectivity for a species could help increase gene flow between different populations and allow a population to shift its range. These two outcomes of the same action may be more or less important depending on the original goal of the project, and each outcome would need different monitoring plans to determine the success of the project. Alternatively, increasing connectivity could result in the spread of a nuisance non-native invasive species. Managers can evaluate trade-offs through processes such as SDM to decide whether such a risk outweighs the benefits of an action and implement early detection monitoring and eradication plans as precautionary strategies (Beaury et al., 2020). Management networks such as the Northeast Regional Invasive Species & Climate Change (RISCC) have readily available resources to help managers identify effective strategies for these concerns. Developing a theory of change is another structured way to articulate goals and objectives to identify monitoring needs (GIZ & CMP, 2020), but even informally sketching goals, objectives and other external forces (such as climate change impacts) can help clarify what to monitor (Thurman et al., 2022).

Monitoring may help inform the rapid and dynamic management actions needed to respond to novel and extreme conditions or shifts in species occurrences outside of historical norms. For example, wide scale monitoring of North Atlantic Right Whales has triggered Dynamic Management Areas to reduce conflicts with shipping and fishing as these highly endangered species have responded to climate change through shifts in seasonal habitat use (Staudinger et al., 2019). In addition, Boult (2023) proposed meteorological forecasting to identify a potential extreme event coupled with monitoring to rapidly respond to a short-term threat and build systems for future responses. In an example of this method, Boult (2023) suggests shading sea turtle eggs if extreme heat is predicted to exceed a pre-identified temperature threshold and conducting close monitoring throughout the forecast period.

Another aspect of dynamic management is changing an overall management strategy due to ecosystem transformation. Monitoring can help determine when an ecosystem has passed a threshold to cue a management change. For example, monitoring the abundance of invasive species at a site might inform management shifts from a Resist to an Accept strategy. On a
broader scale, monitoring on the community level could indicate when novel species interactions pass a threshold from rare to common (Powell et al., 2017). As species shift their ranges, species may hybridize, and monitoring may help assess hybrid offspring fitness and population dynamics (Thompson et al., 2023).

Monitoring to evaluate objectives in climate change is complex as species are simultaneously responding to multiple cues. Therefore, monitoring suites of abiotic drivers at similar spatio-temporal scales at which species occur is critical to understand when, where, and why changes are occurring and to design management strategies that can respond accordingly (Staudinger et al., 2013; 2019). At the habitat level, there may be multiple reasons to protect and manage an area, requiring the assessment of multiple drivers. For example, an area may have complex terrain that provides microclimates or different canopy cover and groundwater inputs that minimize temperature extremes over fine spatial scales (Morelli et al., 2016). Each of these distinct characteristics would need different monitoring objectives to assess the effectiveness of the refugia as climate continues to change (Morelli et al., 2016).

a. Maladaptation

Because many climate adaptation actions are untested, they have the potential to unintentionally create negative impacts, referred to as maladaptation. Therefore, monitoring is critical after an action is taken to track potentially negative social or ecological consequences (Reckien et al., 2023). There is no clear-cut line to determine if an action is adaptive or maladaptive (Reckien et al., 2023). Many actions will have mixed results, or an action can be adaptive initially but as the climate changes, the negative consequences increase. Palik et al. (2022) suggests monitoring for at least five years after implementing forest-assisted migration with regular assessments to identify successes, limitations, and unforeseen outcomes.

Although the potential for maladaptive outcomes is important to consider when planning and implementing adaptation actions, the novel effects of climate change make it unlikely to foresee all unintended consequences, making monitoring a critical component on the planning and implementation process. Ecologically, an adaptation action might be implemented for one purpose at the expense of another goal. For example, when reforesting an area to store carbon, the species used for reforesting could displace native plants and animals (Mullenbach et al., 2022). An action may also create mixed outcomes, as was demonstrated by modeled outputs of flood elements that varied based on different levels of reforestation in the Connecticut River.
Valley (Palmer et al. 2019). Reforestation of both tributaries and the mainstem of the Connecticut River was found to reduce the flood peak, but also increased the number of buildings flooded, particularly upstream (Palmer et al. 2019). Based on these findings, it was recommended that restoration be restricted to the mainstem to balance ecological and social benefits (Palmer et al. 2019).

Additionally, the social outcomes of adaptation could result in winners and losers from different communities (Dana, 2022; Praskievicz, 2023). For example, building green space in an urban environment to help manage increased storm runoff could make a neighborhood more desirable, thereby increasing home values and pushing low-income residents out of the area (Mullenbach et al., 2022).

II. DEMONSTRATIONS TO INTEGRATE CLIMATE ADAPTATION INTO THE SWAP PLANNING PROCESS

A) CREATING CLIMATE-INFORMED SWAP ACTIONS

To fulfill a required SWAP element, states are tasked with creating actions to conserve species or habitats. It is at this stage of action design where climate information can be integrated. We propose a process for developing these climate informed actions that identifies what climate information is known about a species, and then builds off information and resources from previous chapters of this synthesis. An initial step in developing climate-informed SWAP actions could be to identify data gaps and to create a plan to fill them. We suggest a way to address these data gaps by first breaking them into two categories: species with little to no information on how climate change will impact them, or that have a key piece of information missing. For more climate data-rich species, we present an example approach of how to apply the 5Ws approach to develop actions.

Throughout Chapter 2 (Biological Responses) we summarize what information is currently known and highlight data gaps at the species level and for major taxonomic groups. In Chapter 3 (Vulnerabilities) we provide species-specific results from climate change vulnerability assessments (CCVAs). In some extreme cases, species and taxa may lack not only climate information but basic life-history research (e.g. fairy shrimp). For data-poor species, basic
research needs to be conducted before developing climate-informed actions, and this research can be demanding. More subtle data gaps may come up as state agencies apply this synthesis to their own SWAP. For example, when looking at CCVA results in Appendix 3, a species may not have a CCVA result for a particular geography. We outline steps below to handle both species without information and those with a key piece of information missing. These steps, including research needs, are more detailed in Chapter 2.

The second series of examples show how climate informed SWAP actions can be developed for species and habitats that are data-rich. For data-rich species, state agency staff can use the 5Ws (Oakes et al., 2021) as a reflective method for integrating life history research and climate-informed tools to develop climate-informed actions. We give examples for how to do this by adapting actions written by Northeast states for their 2015 SWAPs using the tools detailed in the appendices and in Section 1 of this chapter. Our goal for this section is to provide examples to empower SWAP authors and state personnel to apply their own local and species-specific knowledge to develop relevant climate-informed actions for their state.

1. Actions for species with little to no information

Species are characterized by:

- No literature in biological responses
- No Climate Change Vulnerability Assessment results
- Species with no proxies

ACTION: Develop a plan for building a knowledge base by conducting research and monitoring on how species respond to key climate variables, prioritizing keystone or indicator species

- Examine studies from other similar taxa for examples of where to start
- Organize topical experts around decision support tools and management frameworks to identify most critical areas of knowledge to focus on initially and develop iterative goals to be accomplished over the next 10 years

2. Actions for species with a key piece of information missing

- A CCVA analysis exists but is missing an assessment of a geographic location, or biological responses for some life history traits are available but not others (e.g. literature on morphological responses but not phenological responses)
ACTION: Prioritize place-based or trait-based research and monitoring studies that fill state-specific data gaps and are strategic within the regional distribution of the species. For example, prioritizing conservation in parts of a species' range where they are predicted to thrive, e.g. northern edge.

ACTION: Implement actions that build from what is known

- If a RSGCN has a Climate Change Vulnerability Analysis ranking and determines it is not vulnerable at a certain location, new spatially explicit actions can prioritize habitat protection where the species is least vulnerable and most likely to persist.
- Use downscaled climate projections to evaluate the amount of exposure these populations and habitats are expected to experience over time to determine if currently occupied areas are expected to remain suitable in the future (Chapter 1).
- Determine climate thresholds for these species, such as upper limits of water temperature for aquatic species (Chapter 2).
- Review CCVA results for species or habitats of interest to determine risk sources and levels (Chapter 3).
- Prioritize connectivity and other relevant actions on high exposure areas with short timelines to climate thresholds or tipping points.
- Evaluate the establishment or expansion of corridors from less climate suitable habitats or populations to less exposed regions or buffered climate refugia.

iii. Actions for data rich species

a. Background on the 5Ws process

In the past ten years there has been a notable increase in climate adaptation tools, maps, planning guides and suggested approaches. Many of these resources are easy to access online, are clearly organized and are paired with case studies. However, most of these resources are calibrated to guide adaptation project planning rather than developing discrete actions. This is largely because managers need to incorporate context-specific information about their system of interest. To help SWAP writers develop climate-informed actions that link the landscape of adaptation tools with their local knowledge, we provide an example of how the 5W approach
from Oakes et al. (2022) could enhance selected actions from the 2015 SWAPs, by reflecting on the questions - Where, When, What, Who and Why.

The 5Ws are a standard set of questions used in multiple fields and were adapted by Oakes et al. (2022) to rapidly assess if a conservation project or nature-based solution was climate informed. They used the 5Ws in a climate context as a reflection tool because as ecosystems are pushed outside of historical bounds, management actions may consider modifications to adjust where they take place or when they take place. Managers may also have to change what they do, such as implementing a novel intervention or a traditional intervention in a new place. Using management frameworks can help managers develop goals to better articulate why they are taking certain actions. Additionally, researchers have argued that actions taken to adapt to climate will have winners and losers (Webster et al., 2017). This means it is important to be intentional about who is involved, considered, collaborated with, or consulted when designing actions.

Below we link tools to the 5Ws to demonstrate how to modify older actions (from the 2015 SWAPs) to be more climate-informed or to link actions to larger decision frameworks. We do this in examples representing two SWAP relevant-scales: 1) at the species scale, using the hellbender salamander, and 2) at the habitat scale, using vernal pools. These two illustrative examples were chosen because there was a relatively large amount of relevant information to build on for each case. More information on specific tools mentioned for each of these examples can be found in Section I or Appendix 4.1.

**Figure 4.8.** *The four proposed steps for developing climate informed SWAP actions using the 5Ws.*
b. Vernal pool example

What?

<table>
<thead>
<tr>
<th>2015 Action</th>
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<td>“Site/area management: Manage sites to maintain natural hydrologies” - Rhode Island</td>
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<th>5W Question</th>
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<tr>
<td>What? Are there ways that traditional actions can be modified to be effective at achieving goals under a changing climate? Are there new actions that can achieve goals, or address new or exacerbated challenges caused by climate change?</td>
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<th>Climate Tools</th>
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<tr>
<td>Tools to modify what is being done:</td>
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<tr>
<td>• Non-Forested Wetlands Adaptation Menu (Staffen et al. 2019)</td>
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<td>Adjust wetland systems to cope with altered hydrology. Use climate data to understand projected hydrologic trends for a site and manage the impact of those trends rather than historical conditions (Staffen et al 2019, Strategy 5, Strategy 4).</td>
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How the modified action enhances the climate change context:
Restoration often uses historical benchmarks to return a system to a “natural” state. However, historical conditions may be increasingly difficult to meet with climate-induced changes in precipitation regimes. Climate models can inform restoration projects
with projections of changing and novel precipitation regimes to consider in plans to maintain both historical properties or ecological functions and the suitability of future conditions under different timelines (e.g., 2050, 2100).
Where?

```
“Develop and maintain a list of vernal pools that should be priorities for land protection to benefit SGCN” - Massachusetts
```

5W Question

```
Where?
Are there particularly strategic places or sites to prioritize in implementation, given potential climate change impacts (e.g., work in places that are more or less likely to be impacted, or places where the chances of successful outcomes may be greatest)?
```

Climate Tools

```
Tools to modify where on the landscape vernal pools are best prioritized and protected:
• Vernal pool inundation modeling tool
```

Rewrite

```
Use the vernal pool inundation modeling tool to assess what pools will persist under different climate projections and use that information to develop and maintain a list of vernal pools to prioritize for land protections to benefit SGCN.
```

How the modified action better fits a climate change context:
As climate changes, biodiversity conservation will become more resource intensive. Using the vernal pool modeling tool can identify vernal pools that require more management resources. This can identify trade-offs in resource allocation and delineate networks of vernal pools than can be maintained across the landscape for the least amount of time, money, and effort.
How the modified action better fits a climate change context:

To be effective, invasive species management action timing may need to shift as climate change impacts plant phenology. For example, invasives may emerge earlier and last longer due to warmer spring conditions. Monitoring can inform flexible and adaptive invasive species management to rapidly respond to changing climatic conditions.
### How the modified action better fits a climate change context:

Based on a review of 18,000 papers, Piñeiro et al. (2020) recommend agricultural incentive programs may be most effective if they are well-targeted. Developing well-targeted programs is more complex now more than ever because the impacts of climate change may affect individuals' and communities' relationships with wildlife (Abrahms et al., 2023; Cunsolo & Ellis, 2018). Previous understandings of communities, value systems, and motivations may not persist in the future. The Tools of Change website has a

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<td>“Work in partnership with the Natural Resources Conservation Service to deliver programs that provide cost-share incentives for private landowners to manage their lands to benefit GCN species and their habitats.” -Connecticut</td>
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<th>5W Question</th>
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<td>Who leads design and implementation, and who needs to be involved for actions to be accepted, effective, enduring, and reflective of the needs and diverse values of people and communities? Does climate change affect who benefits or should benefit from actions? Who might be harmed by actions or bear costs?</td>
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<th>Climate Tools</th>
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<tr>
<td>Tools to design effective incentive programs: create effective informational outreach:</td>
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<td>• The Tools of Change website resources, specifically Building Motivation over time</td>
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<td>Work in partnership with the Natural Resources Conservation Service to deliver programs that provide cost-share incentives for private landowners to manage their lands to benefit SGCN species and their habitats. Identify specific actions landowners can take and the motivations and barriers to take those actions. Build an incentive program that builds motivation and reduces barriers.</td>
</tr>
</tbody>
</table>
planning toolkit with resources and links to trainings to help managers develop background research and implement incentive programs that consider climate change information.
Why?

Even with modifications in actions, is there a need to adjust the project goals to be more realistic or feasible as the climate changes (e.g., focus on different targets, or strive for different objectives)?

Tools to adjust project goals to be more realistic or feasible as the climate changes:
- RAD
- RRT

Resisting changes to the vernal pool mosaic for as long as the climatic conditions allow, with plans to transition the ecosystem once climatic conditions no longer support vernal pools. Informed by the RRT framework

How the modified action better fits a climate change context:
Management actions that help ecosystems resist the impacts of climate change allow species to adapt by giving them time to shift their ranges or undergo genetic adaptations. Resisting change also gives managers time to create plans to direct ecosystem transformation when resisting change is no longer feasible. Determining a threshold for when to make management changes from resisting to directing allows managers to thoughtfully transition an ecosystem to support a suite of new and novel species.
c. Hellbender Salamander (Cryptobranchus alleganiensis) example

What?

2015 Action

“Assess feasibility of captive-breeding, head-starting, and/or moving egg masses to reintroduce or expand populations.” - Maryland

5W Question

What?
Are there ways that traditional actions need to be modified to be effective at achieving goals under a changing climate? Are there new actions that will be needed to achieve goals, or address new or exacerbated challenges caused by climate change?

Climate Tools

Tools to find actions to modify captive-breeding and reintroduction programs:
• Terrestrial Wildlife Menu, Adaptive Capacity Menu

Rewrite

Use downscaled climate projections of key climate drivers for species to determine future habitat suitability under different climate projections before translocating or introducing new populations. Specifically translocate individuals with climate adapted genetic traits. Informed by Terrestrial Wildlife Menu Strategy 1

How the modified action better fits a climate change context:
Actions already being implemented, such as captive breeding programs, may need to be adjusted to take advantage of the most up-to-date information or to conduct targeted research to fill knowledge gaps on species’ adaptative capacity to make them more climate-informed.
When?

2015 Action

“Ensure that outdoor recreational activities are not conducted at times or locations that would be detrimental to SGCN.” - New York

5W Question

When?

Does the effects of changing climate increase the urgency of actions that are already being implemented? Would such climate-informed actions need to occur at different times of the year to be effective as the climate changes?

Climate Tools

Tools to identify how to shift the timing of recreation closures:

- Climate Data Chapter 1, Outdoor Recreation Menu

Rewrite

Model changes of the timing of key life history events and timing of recreational events due to changing climate (e.g. warmer spring weather) to identify areas of temporal or spatial overlap. Ensure that outdoor recreational activities are limited at times or locations that would be detrimental to SGCN, by enforcing flexible closure dates based on changing conditions. Informed by Outdoor Recreation Menu, Strategy 3

How the modified action better fits a climate change context:

Traditional fixed closure dates may be out of sync with climatic trends. Flexible closure dates can track climate changes or respond to extreme events within critical life history windows.
57

Who?

“Publish eastern hellbender and mudpuppy educational materials in the PA Angler & Boater for distribution.” -Pennsylvania

Who?
Who leads design and implementation, and who needs to be involved for actions to be accepted, effective, enduring, and reflective of the needs and diverse values of people and communities? Does climate change affect who benefits or should benefit from actions? Who might be harmed by actions or bear costs?

Tools to identify how to engage communities for enduring actions:
- Words Matter

Use the specific results for anglers and boaters in Words Matter to develop education materials that align with their attitudes, values and priorities. Circulate materials to Department of Transportation staff for their consideration in maintenance and development plans and build cross department collaboration.

How the modified action better fits a climate change context:
Educational materials are an important first step to engaging communities and other agencies. Using resources like **Words Matter: Determining How to Engage the American Public Through the Language of Conservation** (see Appendix 4) can help increase the impact of these materials. Additionally, as populations decline or need more resource-intensive conservation, engaging local
community members in monitoring through citizen science campaigns, for example using iNaturalist or National Phenology Network's tool Nature's Notebook, can help reduce the burden on agency staff.
Where?

“Protect migration corridors and install road crossing tunnels where applicable to mitigate road mortality.” -New York

Where?
Who leads design and implementation, and who needs to be involved for actions to be accepted, effective, enduring, and reflective of the needs and diverse values of people and communities? Does climate change affect who benefits or should benefit from actions? Who might be harmed by actions or bear costs?

Tools to identify where to protect road crossings:
- Biologic responses Chapter 2, Climate data Chapter 1, Terrestrial Wildlife Menu, Staying Connected Initiative

Rewrite

Protect migration corridors and install road crossing tunnels where applicable to mitigate road mortality. Evaluate key current and future road crossings to ensure they are built to withstand extreme precipitation and aquatic connectivity will be maintained under future conditions. Use known biological responses of Hellbenders to climate change and climate modeling maps to mitigate road mortality in both the known species range and areas the species might shift into.

How the modified action better fits a climate change context:
Managing road crossings will be particularly important as species shift their geographic ranges or phenology. It will be important to both prioritize areas in a species’ historical range as well as to create landscape connectivity based on future projections of habitat suitability and climate refugia to facilitate range shifts and maximize persistence.
Why?

Even with modifications in actions, is there a need to adjust the project goals to be more realistic or feasible as the climate changes (e.g., focus on different targets, or strive for different objectives)?

Tools to adjust project goals to be more realistic or feasible as the climate changes:
- RAD
- RRT

As climate changes and conservation takes more resources it will be important to assess the status of a species within its range, and triage management for different populations. Specifically, resisting changes to maintain some populations. For example, increasing genetic diversity in trailing edge or remnant. Accepting some populations may be lost. Or directing change such as translocation outside a population's historical range. Informed by RAD and Thurman et al. 2021, S5.

How the modified action better fits a climate change context:
As climate change and conservation requires more resources, it will be essential to triage species, specifically increasing genetic diversity in trailing edge or remnant populations if those populations are important for maintaining the evolutionary potential of the species (Thurman et al. 2021, S5).
B) CLIMATE AMPLIFIER VIGNETTES

Both direct and indirect impacts of climate change need to be accounted for when developing management plans. Indirect impacts can be difficult to identify and plan for because climate change will transform ecological and social systems, creating vast and nuanced outcomes. One way to target indirect effects through management plans is to evaluate the ways climate change can interact and amplify non-climate threats to species. Additionally, managing at the threat level can facilitate developing actions that benefit a suite of species, including co-occurring data-poor species. A survey of taxa team experts conducted by Terwilliger Consulting Inc in 2019, identified non-climate threats (hereafter "threats") and climate amplifiers to RSGCN through a series of targeted questions (TCI & NEFWDTC 2023b). Using the results of this study, we grouped species with an identified shared threat and a known climate amplifier to develop three case studies illustrating actions that can simultaneously address both climate and non-climate threats. These examples list the species identified by the taxa teams, build out a problem statement to explain how the climate impact amplifies the threat, and list proposed solutions. Additionally, we map the proposed solutions to approaches from the Terrestrial Wildlife Menu and the Tribal Adaptation Menu to demonstrate alternative ways these approaches can be adapted to inform SWAPs. This section outlines a threat-based framework for identifying and creating solutions to interactive and indirect multi-stressor effects.

i. The effect of housing and urban areas amplified by hydrology/precipitation

RSGCN

- Bog Turtle (*Glyptemys muhlenbergii*)
- Bridle Shiner (*Notropis bifrenatus*)
- Swamp Darter (*Etheostoma fusiforme*)
- Western Pirate Perch (*Aphredoderus sayanus gibbosus*)

Habitat

- Bog Turtle- Shallow freshwater wetlands with slow moving water, open canopy areas like fens
• Bridle Shiner, Swamp Darter and Western Pirate Perch- Slow moving freshwater, such as pools backwater and swamps

Key points from taxa team experts

Increased impervious surface areas due to expanding urban areas, combined with soil compaction, vegetation removal, and piping leads to more runoff. This additional runoff creates flashy flows and can add increased pollution and sedimentation in streams, thus degrading aquatic habitats. In the Northeast, climate change is predicted to cause more extreme precipitation events which amplify runoff and resulting erosion and pollution. These two forces will degrade habitat and stress freshwater species. Additionally, a taxa team expert noted: dams and other water management infrastructure can make streams warmer, exacerbating warming water due to climate change.

Problem statement

All four RSGCN identified at the intersection of development (non-climate stressor) and changes in hydrology (climate amplifier) share similar habitat, slow moving freshwater, and for this vignette we focus on habitat management actions. All of the identified species sensitive to habitat degradation, are relatively rare and often occur in isolated populations (Geneva et al., 2018; Stratmann et al., 2016). This suggests that a few habitat-degrading events could wipe out entire populations for any of these species. Bog Turtles are especially sensitive and threatened by development because they are habitat specialists, and their habitat --open wetlands-- often overlaps with productive agricultural lands that are not protected (USFWS, 2001; Stratmann et al., 2016). While Bog Turtles have been sighted in human-altered habitats, such as agricultural ponds, urban development combined with changes in hydrology could greatly reduce their remaining available habitat (USFWS, 2001). Increased development is likely to lead to habitat fragmentation for all RSGCN, but put Bog Turtles at even higher risk due to the added threat of vehicle strikes if they need to cross roads to reach suitable habitat (USFWS, 2001). Additional fragmentation could also be detrimental to Bridle Shiner populations, due to low genetic diversity and adaptive capacity throughout their range (Geneva et al. 2018).

Urban streams often have increased high flow events, altered geomorphology, increased dissolved solids and insecticides, and reduced biodiversity across trophic levels (Walsh et al. 2005). This collection of attributes is known as urban stream syndrome (Walsh et al. 2005, Paul...
While urban stream syndrome is well understood, research on how increased development, including the downstream effects, specifically affects the four highlighted RSGCN is uneven.

Climate change may exacerbate urban stream syndrome in the Northeast because precipitation will come less frequently but in more extreme amounts (see Chapter 1) leading to increased runoff, erosion, and pollution. Additionally, the runoff may pick up heat from the impervious surfaces increasing the temperature of the water bodies it flows into (Waite et al., 2019). Pollutants can change the invertebrate and plant assemblages in aquatic environments by killing off sensitive species that are replaced by generalist species. These amplified effects of urban stream syndrome could increase habitat-degrading events, creating a larger threat for these RSGCN.

Solutions

The disturbance caused by urban and housing development, and legacy impacts of development directly threaten these four RSGCN. As climate change exacerbates urban stream syndrome, the downstream effects of increased sedimentation, pollution and heat could affect populations of these species that occur both inside and outside of urban areas (Klemens, 2001; Tang et al., 2020). Additionally, urbanization is expected to increase throughout the Northeast as climate change pushes people away from the coasts and into more temperate interior regions or from other parts of the United States and world into the Northeast. Larger scale approaches to reduce the negative effects of urbanization on stream ecosystems could be working towards both protecting core areas of habitat, promoting stormwater control measures that reduce the landscape effects of urban streams, and engaging communities to restore or protect urban streams. At the species level, translocating individuals or populations of at-risk species may be a potential option to maintain viable populations in undisturbed habitat.

Solution 1: Identify and protect core habitat

The 2023 The Nature Conservancy Conservation Status of Natural Habitats assessed development in watersheds and provided tools and maps for prioritizing land and stream conservation and connectivity through protected areas and reduced disturbance (Anderson et al., 2023).
Solution 1 corresponds to:

- **Terrestrial Wildlife Menu Strategy 8** - Restore, and maintain sources of food, water, and cover as components of habitat.
- **Terrestrial Wildlife Menu Strategy 10** - Establish and enhance protected areas of habitat reserves.
- **Tribal Adaptation Menu Strategy 4** - Sustain fundamental ecological and cultural functions.
- **Tribal Adaptation Menu Strategy 8.4** - Establish protected areas to maintain ecosystem and cultural diversity.

Solution 2: Reducing non-climate stressors through improved Stormwater Control Measures

Walsh et al. (2016) developed five principles to develop stormwater control measures (SCMs) that protect stream ecosystems and provide a foundation for managers to coordinate with local governments or other entities to improve stormwater infrastructure. While SCMs often prioritize public health and flood mitigation, the principles in Walsh et al. (2016) focus on the environmental conditions of downstream nonurban catchments. The authors outline how SCMs can help reduce flashy flows and delivery of pollutants to downstream areas by restoring and protecting surrounding areas, striving to achieve pre-development hydrological conditions, and reducing impervious surfaces and untreated runoff.

Solution 2 corresponds to:

- **Terrestrial Wildlife Menu Strategy 9.5** - Establish or redesign infrastructure to protect habitat from anticipated climate impacts.
- **Tribal Adaptation Menu 5.4** - Reduce negative impacts from anthropogenic disturbances.

Solution 3: Engaging communities

Urban communities have a vital interest in stream restoration to increase ecosystem services, aesthetic value, and recreation potential. Importantly, restored urban streams can help reduce negative downstream impacts (Konrad & Booth 2005). Engaging communities and
aligning restoration plans with community values is critical for success (Metcalf et al., 2015). For example, social and governance resistance are known barriers to developing new, ecologically focused, stormwater control measures (Walsh et al. 2016). Scoggins et al. (2022) recommends four actions managers can take to effectively engage communities in urban stream restoration projects including actions so mangers can first understand community perspective and then meaningfully collaborate with community members throughout the life of a restoration project (Scoggins et al. 2022). While these guiding principals were developed for urban watersheds, they are generally transferable to other socioecological systems. These engagement points can also serve as opportunities to introduce and educate the community to the relevant climate data and tools needed to fully address threats to RSGCN.

Solution 3 corresponds to:

- **Tribal Adaptation Menu Strategy 1.1** -Consult cultural leaders, key community members, and elders.
- **Tribal Adaptation Menu Strategy 1.2** - Consider mindful practices of reciprocity.
- **Tribal Adaptation Menu Strategy 1.3** - Understand the human and landscape history of the community.

Solution 4: Translocation

Translocation could be tactic to increase genetic diversity in some populations and move individuals from unprotected areas to protected areas that have been evaluated for their future climate suitability using downscaled climate projections. Dresser et al. (2018) recommend translocation for some southern populations of Bog Turtles that were small and isolated, after conducting a genetic survey. While northern populations of Bog Turtle have relatively more protections and may be less imperiled than southern populations, assessing the genetics in the northern population could be a first step to establish baseline data followed by monitoring for inbreeding and bottlenecks to determine if translocation is needed in the future. Thompson et al. (2023) provides guidance for reducing the risk of translocation. Bridle Shiners could also be
considered for translocation and Jensen (2012) suggested areas with dense spring vegetation with 
slow water flow were suitable destination habitats.

Solution 4 corresponds to:

- **Terrestrial Adaptation Menu Strategy 1.9** - Restore genetic diversity in isolated or 
inbred populations
- **Terrestrial Adaptation Menu Strategy 2.1** - Translocate individuals or populations to 
habitat within the existing range that was formerly occupied and remains suitable
- **Tribal Adaptation Menu Strategy 11.2** - Establish or encourage new mixes of local 
beings and/or bakaan ingoji gaa-ondaadag (Anishinaabemowin word for non-local 
beings) expected to do well under future conditions to meet future needs.
- **Tribal Adaptation Menu Strategy 11.4** - Seek out and share traditional and cultural 
knowledge of potential new beings from Tribal communities where these beings are 
native.

*ii. The effect of renewable energy amplified by habitat change*

**RSGCN**

- Silver-haired Bat (*Lasionycteris noctivagans*)
- Eastern Red Bat (*Lasiurus borealis*)
- Hoary Bat (*Lasiurus cinereus*)
- Tri-colored Bat (*Perimyotis subflavus*)

**Habitat**

- Silver-haired Bat, Eastern Red Bat, Hoary Bat, Tri-colored Bat - Forested Woodland, 
  Developed Areas, Riparian Floodplain, Lakes and Ponds, Great Lakes, Grassland, Glade 
  Barren Savanna, Non-Tidal Wetlands, Tidal Wetlands, Rivers and Streams, Shoreline 
  (except for Hoary Bat), Marine Nearshore (except for Tricolored Bat) and Shrubland 
  (except Hoary Bat)
- Eastern Red Bat - Agricultural Cropland
- Tricolored and Silver-haired Bats - Cliff Talus
Key points from taxa team experts

Increased wind energy development will potentially decrease bat RSGCN habitat. Additionally, changes in phenology of bats may make it difficult or impossible to create a wind curtailment schedule to reduce bat mortality at wind farms.

Problem statement

Bat fatalities caused by wind energy installments can increase as migration patterns and habitat use shifts in response to climate change. These fatalities could be consistent across species and habitat (e.g. terrestrial and offshore wind development), and so we take a threat based approach for this vignette. Wind energy installments have the potential to greatly impact bat populations by increasing fatalities by either directly striking bats, or through barotrauma (Sjollema et al., 2014; Friedenberg & Frick, 2021). Barotrauma is a sudden pressure change that damages air-containing organs (Sjollema et al., 2014). Friedenberg and Frick (2021) modeled the effect of terrestrial-installed wind energy on Hoary Bat populations under two build-out scenarios in the U.S. and Canada; mean declines were estimated at 32–70% and 17–50% under high and low build-out scenarios by 2050, respectively. Declines were projected to vary depending on bat population growth rate and wind energy development. One estimate showed a population with an annual growth rate of 18% was projected to decline and risk extinction in less than 40 years (Friedenberg & Frick, 2021).

Bats have also been observed to use offshore ocean habitats, both during migration periods and throughout the summer, putting them at risk for interactions with a rise in offshore wind developments (Dowling, 2018, Dowling and O’Dell 2018, Sjollema et al., 2014). Taxa team experts also noted that bats will likely lose habitat due to increased wind energy installments. Most bat fatalities with wind energy developments occur between mid-July and October, coinciding with when bats migrate (Hayes et al., 2015). Studying bat species that respond to climate change through shifts in migration patterns will help to inform and adapt wind energy curtailment programs. Additionally, the high rate of mortalities caused by wind turbines could reduce genetic diversity, lowering adaptive capacity to cope with environmental change. Reducing bat and other wildlife fatalities with wind energy installments is beneficial to wildlife managers and the wind energy industry to reduce population declines. Extreme population declines could trigger protective policies that halt new development or operation schedules.
Solutions

Solution 1: Proactive curtailment when bat activity is high

Proactively curtailing terrestrial wind energy production to wind speeds of 5 m/s can achieve similar outcomes as smaller build-outs to maintain bat population viability and potentially reduce future conflict as curtailment was found to be more socially and politically feasible than reducing wind infrastructure (Friedenberg & Frick, 2021). However, cutoff speeds may be different for offshore wind developments because bats may be active during higher-than-optimal wind speeds so they can return and find terrestrial roosting sites (Sjollema et al. 2014). Models of curtailing offshore wind speeds projected minimal losses of energy production (Dowling 2018), pointing to a balanced solution for both wildlife management and renewable energy expansion.

Solution 2: Acoustic deterrents

Acoustic deterrents are a possible solution, but more research is needed on their efficacy before adopting wider use (Frick et al., 2020)

Solution 1 and 2 correspond to:

- **Terrestrial Wildlife Adaptation Menu -Strategy 7.1**: Anticipate and manage conflict from increasing populations, range expansions, or changing behaviors.

- **Tribal Adaptation Menu -Strategy 13**: Design and modify infrastructure and access to match future conditions and community needs.
  - 13.2. Incorporate natural or low impact development into designs.

- **Terrestrial Wildlife -7.2** Manage conflict associated with societal adaptations to climate change

- **Tribal Adaptation Menu -5.4**. Reduce negative impacts from anthropogenic disturbances

Solution 3: Research on life-history of bat species, and biotic responses to climate change

Many bat species are data-deficient, so continued monitoring is important to understand how bats adjust to climate change and interact with wind installments (Frick et al., 2020, Hayes
et al., 2015). This is particularly true as bat species expand their range (Loeb & Winters, 2013) or change their phenology (Jonasson & Guglielmo, 2016).

- **Tribal Adaptation Menu Strategy 2**: Learn through careful and respectful observation (gikinawaabi).
- **Tribal Adaptation Menu 2.1**: Learn from beings and natural communities as they respond to changing conditions over time.
- **Tribal Adaptation Menu 7.4**: Seek out or share traditional and/or cultural knowledge to inform management of sensitive or at-risk beings or communities.

iii. The effect of invasive or non-native species amplified by changing temperature regimes

**RSGCN**
- Summer Sucker (*Catostomus utawana*)
- Dwarf Wedgemussel (*Alasmidonta heterodon*)
- Brook Floater (*Alasmidonta varicosa*)
- Ipswich Sparrow (*Passerculus sandwichensis princeps*)
- Wood Thrush (*Hylocichla mustelina*)
- Big Sandy Crayfish (*Cambarus callainus*)

**Habitat**
- Summer Sucker - Summer Suckers are endemic to the Adirondack region of New York and live in lakes, creeks and small rivers with rocky pools and runs.
- Dwarf Wedgemussel - Dwarf Wedgemussels are in streams, big rivers and riparian floodplains from New Hampshire to North Carolina.
- Brook Floater - Brook Floaters are found in streams with clear water, low to moderate and stable substrates. Their range spans from eastern Canada to Georgia.
- Ipswich Sparrow - Ipswich Sparrows are a subspecies of the Savannah Sparrow that exclusively breeds on Sable Island Nova Scotia and winters in sand dunes along the eastern U.S.
• Wood Thrush - Wood Thrushes breed in deciduous and mixed forest in the Northeast United States. They nest in the understory and forage for invertebrates in leaf litter (Lambert et al. 2017).
• Big Sandy Crayfish - Big Sandy Crayfish are found in pool and riffle streams with large boulders in the Appalachian Region of Virginia, West Virginia, and Kentucky.

Key points from taxa team experts

Across taxa and habitats, warmer temperatures will facilitate invasion of new species and increase abundance of current invasive species. Invasive species may outcompete native species or harm individuals as pests or pathogens.

Problem Statement

RSGCN identified at the intersection of the invasive species and warming temperatures are all impacted in different ways by this threat and amplifier interaction. The taxa teams did not specify which invasive species or how an invasive species would impact these RSGCN; therefore, additional research was used to explore their concerns further. Below we present examples of how prominent invasive species currently impact or could impact RSGCN within their respective habitats and solutions to the threat amplifier interaction in a slightly different format than the previous two vignettes (Table 4.8).

Table 4.8. RSGCN identified by taxa team experts as threatened by invasive species and increasing temperatures, linked to information on prominent non-native threats within their respective habitats.

<table>
<thead>
<tr>
<th>Species</th>
<th>Most prominent impact by invasive species</th>
<th>Citations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Big Sandy Crayfish</td>
<td>The Fish and Wildlife Service’s recovery outline for Big Sandy Crayfish and Guyandotte River Crayfish noted that streams with Japanese Knotweed (<em>Fallopia japonica</em>) could be altering the habitat for these two crayfish species and causing their decline. Japanese Knotweed has shallow roots and dies back in the winter; these life history properties often lead to increased erosion and sedimentation along stream banks.</td>
<td><a href="https://www.fws.gov">U.S. Fish &amp; Wildlife Service Species Profile</a></td>
</tr>
</tbody>
</table>
Rusty Crayfish (*Orconectes rusticus*), is an aggressive species invading streams outside of their native range in the Ohio River Basin and has been identified as a threat to native crayfish species in the Northeast (Olden et al., 2006). As temperatures warm, and river systems become more habitable for Rusty Crayfish, they could impact Big Sandy Crayfish. However, more research is needed to determine how these two species could interact.

<table>
<thead>
<tr>
<th>Species</th>
<th>Threats/Impacts</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brook Floater</td>
<td>Non-native species can alter habitat and compete for limited resources, specifically <em>Corbicula fluminea</em> and zebra mussels (<em>Dreissena polymorpha</em>) are cited as potential species that could negatively affect the Brook Floater population.</td>
<td><a href="#">Species Status Assessment Report for the Brook Floater</a></td>
</tr>
<tr>
<td>Summer Sucker</td>
<td>Predation and competition by invasive species like Largemouth Bass (<em>Micropterus salmoides</em>), Smallmouth Bass (<em>Micropterus dolomieu</em>), and Yellow Perch (<em>Perca flavescens</em>) are likely causes of species loss or decline.</td>
<td><a href="#">New York Species Assessment</a></td>
</tr>
<tr>
<td>Wood Thrush</td>
<td>It is unclear what specific invasive species are threatening Wood Thrushes. However, the New Hampshire Wildlife Action Plan (2015) suggested that insect pests can alter tree compositions and litter composition within forest habitat (Hornbach et al., 2021).</td>
<td><a href="#">Lambert et al., 2017</a></td>
</tr>
<tr>
<td>Dwarf Wedgemussel</td>
<td>The New York Species assessment found that invasive zebra mussels pose a threat to Dwarf Wedgemussel populations and are a general threat to native freshwater mussel species. Didymo (<em>Didymosphenia geminata</em> or rock snot), a filamentous diatom, can alter mussel habitat by forming mats that smother stream bottoms.</td>
<td><a href="#">Species Assessment for Dwarf Wedgemussel</a></td>
</tr>
<tr>
<td>Ipswich Sparrow</td>
<td>Unknown</td>
<td></td>
</tr>
</tbody>
</table>

**General Impacts**

Beyond the impacts to the focal species mentioned above, non-native biological invasions can cause profound ecological transformations and, in some cases, profound harm to ecological
Lopez et al. (2022) conducted a meta-analysis to understand the effects of global environmental changes (GECs), specifically drought, nitrogen addition and warming, and invasive species. They found that while the combined effects of biological invasions and GECs are greater than GECs alone, it is no worse than invasions alone, meaning that biological invasions have outsized impacts compared to GECs.

These ecological impacts may increase as climate change facilitates non-native invasive species in several ways. Earlier spring due to climate warming allows invasive plants (whose growth is often queued by temperature rather than light) to emerge earlier and outcompete native plants (Willis et al., 2010; Wolkovich & Cleland, 2011). Longer growing seasons and warmer temperatures also support the survival of a greater diversity of insect pests and respective number of broods that can be produced in a season. Additionally, increasing magnitude and/or frequency of extreme events due to climate change create more disturbance opportunities for invasive species to establish and thrive (Díez et al., 2012). Another impact of climate change, increasing atmospheric CO₂, allows invasive plants to become more competitive (Sorte et al., 2013) and could reduce the efficacy of chemical herbicides (Ziska, 2003). Invasive species are actively shifting their ranges with climate change and the Northeast is a hotspot for future invasion (Allen & Bradley, 2016). For example, climate change is projected to make one or more northeastern states (from Kentucky to Maine) newly climatically suitable for 291 invasive plants, of which 74 have been assessed as having ecological impacts that reduce native biodiversity (Coville et al., 2021; Rockwell-Postel et al., 2020).

Solutions

By looking at the RSGCN sitting at the intersection of the threats of invasive species and changing temperature, it is apparent that climate change can lead to greater invasive species impacts and novel invasives. However, there are standard conservation actions that, when implemented in a climate-informed way, can reduce the impact of current and future invasive species across taxa and impacts.

Below, we tag solutions suggested by Beaury et al. (2020) with corresponding approaches from the Terrestrial Wildlife Menu and the Tribal Adaptation Menu. We link these context-specific solutions to other management resources to demonstrate how the broader
approaches and strategies from these resources can be adapted to species and threat-specific context.

**Prevention**

The most effective way to avoid negative impacts from invasive species is to prevent invasions (Bradley et al., 2023; Keller et al., 2007). Managers can curtail the spread of non-native invasives on several scales. Local policies could restrict ornamental plant sales, so that plants that are known or likely invaders cannot be sold. Or, developing restrictions on the types of bait used by anglers can prevent the spread of invasive crayfish into new waterways. Critically, cross-boundary policies and collaboration are important as non-native invasives spread across political boundaries (Bradley et al., 2023; Rockwell-Postel et al., 2020). Preventing the spread of non-native invasives can be accomplished not only by restricting negative behaviors but also by fostering positive behaviors. Managers can encourage people to plant native plants rather than non-natives, or create stations to encourage boaters to clean and dry their boats to prevent the spread of mussels (Bradley et al., 2023). Future-oriented planning activities, scenario planning, or horizon scanning could also help managers plan for potential invaders. Prevention can also happen within the context of climate interventions such as ensuring materials like gravel used in restoration are weed free, making sure carbon capture tree planting initiatives are using appropriate trees, or that plants grown for biofuel are not invasive (Bradley et al., 2023; Rockwell-Postel et al., 2020).

**Table 4.9.** Solutions for reducing the threat of invasives species and the correlating strategy from the Terrestrial Wildlife Adaptation Menu and the Tribal Adaptation Menu.

<table>
<thead>
<tr>
<th>Solutions</th>
<th>Climate Change Modification</th>
<th>Terrestrial Wildlife Adaptation Menu</th>
<th>Tribal Adaptation Menu</th>
</tr>
</thead>
<tbody>
<tr>
<td>Monitoring for new invasive species</td>
<td>Monitoring can look for species spreading from southern regions or species that are currently not invasive but could be under the right conditions,</td>
<td>Strategy 4: Manage interspecific and biotic interactions -Detect and remove non-native invasive species</td>
<td>Strategy 2: Learn through careful and respectful observation (gikinawaabi). -2.1. Learn from beings and natural communities as they respond to changing conditions over time</td>
</tr>
<tr>
<td>Rapid Response</td>
<td>Strategy 3: Support Tribal engagement in the environment. -3.3. Establish, maintain, and identify existing inventory and monitoring programs.</td>
<td>Strategy 4: Manage interspecific and biotic interactions -Detect and remove non-native invasive species.</td>
<td>Strategy 5: Reduce the impact of biological and anthropogenic stressors -Maintain or improve the ability of communities to balance the effects of <em>bakaan ingoji gaa-ondaadag</em> (non-local beings).</td>
</tr>
<tr>
<td>Cross Boundary Partnerships</td>
<td>Cross boundary partnerships can help prevent the spread of invaders, as species spread into new regions learning how other managers in different regions responded to certain invasive species can help managers hear “lessons learned” and respond to species more effectively.</td>
<td>Strategy 3: Support Tribal engagement in the environment.-3.6. Participate in local- and landscape-level management decisions with partner agencies.</td>
<td></td>
</tr>
<tr>
<td>Mitigation</td>
<td>Mitigation actions may need to shift as climatic patterns shift. For example, spraying for invasives may need to happen earlier in the year due to warming springs leading to sooner emergence of species.</td>
<td>Strategy 4: Manage interspecific and biotic interactions -Detect and remove non-native invasive species.</td>
<td>Strategy 5: Reduce the impact of biological and anthropogenic stressors -Maintain or improve the ability of communities to balance the effects of <em>bakaan ingoji gaa-ondaadag</em> (non-local beings).</td>
</tr>
<tr>
<td>Restoration</td>
<td>Restoration and planting native species can help reduce the ability of invasives to outcompete native species.</td>
<td>Strategy 8: Restore, and maintain sources of food, water, and cover as components of habitat.</td>
<td>Strategy 4: Sustain fundamental ecological and cultural functions. Strategy 8: Maintain and enhance</td>
</tr>
<tr>
<td>Engagement</td>
<td>Public engagement has always been a crucial component for reducing the spread of invasive species. These efforts will be continuously important to help managerial capacity monitor for new species, and to help prevent spread as many potential vectors will increase, for example a longer boating season.</td>
<td>Strategy 13: Engage human communities in wildlife conservation</td>
<td>Strategy 1: Consider cultural practices and seek spiritual guidance.</td>
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<tr>
<td></td>
<td>Managers and the public in the United States have had to cultivate some acceptance of invasive species on the landscape. As climate change increases the number and abundance of invaders it may be increasingly important to accept species or communities of invasive species to prioritize efforts.</td>
<td>Strategy 12: Intentionally choose to take no action</td>
<td>Strategy 2: Learn through careful and respectful observation (gikinawaabi).</td>
</tr>
<tr>
<td></td>
<td>Restoration that resists climate impacts, for example restoration that allows for stable hydrologic regimes even with changing precipitation, will help native communities resist invasions</td>
<td>Community and structural diversity.</td>
<td>-2.1. Learn from beings and natural communities as they respond to changing conditions over time.</td>
</tr>
</tbody>
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CHAPTER 5: CLIMATE ADAPTATION CASE STUDIES

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I. INTRODUCTION

In this chapter, we detail four case studies that demonstrate how managers faced novel interactions, responded to current and future threats, and adapted their management actions to mitigate threats in the face of climate change. These case studies take place in different locations, on different scales, and in different systems. However, they confronted the legacies of past land management decisions and policies interacting with new climate regimes, creating novel outcomes. In each case study, managers had to address these novel outcomes while balancing multiple objectives. Additionally, managers looked to the future to identify potential risks and uncertainties, and created plans and implemented actions that would address these risks under various conditions.
Across different perspectives, scales, and points in the decision-making process, these case studies demonstrate the complexity of being a natural resource manager faced with the impacts of climate change. They also demonstrate the creative and responsive ways managers adapt to these impacts. In Case Study 1, the National Park Service (NPS) used scenario planning to integrate climate adaptation into future restoration plans for vulnerable species, habitats, and infrastructure, including at Maine’s Acadia National Park. In Case Study 2, a forest stewardship plan for a Wildlife Management Area in New Jersey was evaluated for its benefit to both wildlife conservation and forest carbon in light of climate impacts. In Case Study 3, managers across the Northeast responded to an indirect climate threat and its impacts on forest habitat, adapting to the unexpected resurgence of an invasive forest pest when a preexisting biological control became less effective due to unusual drought. For the final case study (Case Study 4), the Michigan Department of Natural Resources took lessons after an extreme rain event and changed how they evaluated water control structures and their impacts on aquatic habitat. Each case study is based around input from key stakeholders, and individual bylines indicate the respective roles of these managers and synthesis authors in writing the text. In the face of uncertainty and complexity, these managers were all able to use their resources to implement future-oriented actions and benefit vulnerable species and their habitats.

I. CASE STUDY 1: NATIONAL PARK SERVICE SCENARIO PLANNING  
Contributed by Amanda Babson, National Park Service

A) ADAPTATION PLANNING

The NPS has been applying scenario planning to support climate adaptation in various ways. Acadia National Park held a scenario planning workshop in 2015 (Star et al. 2016), which supported incorporating climate adaptation into subsequent planning efforts, including restoration projects at two sites within the park, Cadillac Mountain and the Great Meadow. By restoring natural hydrology to the Great Meadow, with awareness of increasing heavy precipitation events, aquatic organism passage is likely to improve, and flooding impacts to trails and other infrastructure are expected to decrease. The scenario planning was embedded in a larger planning effort, Acadia’s Resource Stewardship Strategy (NPS 2023). Stewardship goals were set for priority natural and cultural resources, including At Risk Species and Fauna. An
example of a goal for Fauna included *Improve Management planning and responses to threats, including climate change, to wildlife and/or wildlife habitat*. A high-priority activity for At-Risk Species included *Conduct workshop to prioritize species and management activities given climate scenarios using the Resist-Accept-Direct framework*. The scenario planning process spurred discussions about which goals would be achievable under scenarios of major change, and the distinctions between what was possible for short-term goals versus long-term goals.

**B) CLIMATE IMPACTS**

For other parks that have not had a scenario planning workshop, NPS supports a simplified process for incorporating climate scenarios into the Resource Stewardship Strategy (Runyon et al. 2021; Schuurman et al. 2023). These are simplified in the Northeast to a Hot-Wet (most change) and a Warm-Damp (least change) scenario, providing a plausible, divergent range of temperature, precipitation, and sea level rise projections. For George Washington Birthplace National Monument’s Resource Stewardship Strategy (NPS 2018), managers included a goal for Federally Listed Species and Candidate Species of *Sustained population viability of federally listed species within potential range, considering climate change*, but did not have any climate specific activities for that goal. Different levels of scenario planning lead to different tangible outcomes, but parks of all sizes and staff capacities are moving towards incorporating climate scenarios into their resource management, including for wildlife.

**II. CASE STUDY 2: SPARTA MOUNTAIN WILDLIFE MANAGEMENT AREA, TERRESTRIAL WILDLIFE ADAPTATION MENU**

*Contributed by Sharon Petzinger, New Jersey Department of Environmental Protection*

**A) BACKGROUND**

Sparta Mountain Wildlife Management Area (SMWMA) is owned by the New Jersey Department of Environmental Protection’s Fish and Wildlife to maintain and manage habitats for a diversity of wildlife species and wildlife-related recreation. SMWMA is 3,461 mostly forested acres located in the Highlands Region of Sussex County, New Jersey. This property has a long history of disturbance: it was mined for iron ore and limestone in the late 1800s to early 1900s; the area was deforested, and large quantities of soil and bedrock were excavated and transported
via rail to other parts of the property. These actions altered the topography, stream flow and wetlands in the area.

The forests on SMWMA today consist mostly of middle-aged (70-100 years old) closed-canopy oak-hickory forests undergoing mesophication, the change in environment that occurs when fire is removed or suppressed in an ecosystem, with an increase in understory vegetation comprised of northern hardwood species such as sugar maple, red maple, black birch, and American beech. Regeneration harvests have been implemented on about 92 acres in this forest type so far, resulting in the regeneration of tree species such as tulip poplar, black cherry, and sometimes aspen (bigtooth and quaking) and gray birch, in addition to oaks and hickories. There are wetland and vernal pool habitats on the site. About 6% of the property contains mixed upland forests dominated by eastern hemlock (*Tsuga canadensis*). Many of the hemlocks are infected/dying from the Hemlock Woolly Adelgid (*Adelges tsugae*), and much of this forest type is rated as an extreme fuel hazard for wildfires.

About half of SMWMA was purchased with a US Forest Service Forest Legacy grant which requires a natural resource plan. The first forest stewardship plan for SMWMA was approved in 2009. A revised plan, which is currently being implemented on SMWMA, was approved in 2017 and expires in March 2027 (NJDEP, 2017). Some local organizations and members of the public expressed concerns about the forestry activities conducted in the 2017 forest stewardship plan, so an addendum was created in 2021 to help address those concerns, which included limiting the size and types of forestry prescriptions implemented on SMWMA.

**B) CLIMATE IMPACTS**

The most important climate impacts to SMWMA assessed include increased temperatures, changes in precipitation patterns (more rain in winter, less rain in summer), and drier soil conditions during the growing season (Whitehead et al., 2023). These changes then translate to impacts on forest and wetland habitats, including increased stress on northern hardwood tree species, increases in tree pests, prolonged flooding followed by rapid drying of vernal pools and wetlands, and increased wildfire hazard, especially in areas where there is excessive hemlock die-off. Other impacts unrelated to climate change include mesophication, pests already impacting the forests (Hemlock Woolly Adelgid, Emerald Ash Borer, Spongy Moth), limits on funding and capacity, and the politicization of forest management, including a
push to limit forest management, in both acreage and treatment type, in the state to maximize local forest carbon stocks.

C) ADAPTATION PLANNING

As a result of the impacts described above, combining wildlife conservation with climate adaptation became the focus of the SMWMA forest stewardship plan. NJ Fish and Wildlife’s Endangered and Nongame Species Program (ENSP) manages the implementation of SMWMA’s forest stewardship plan, and sent a biologist to attend the Northern Institute for Applied Climate Science (NIACS) courses for Terrestrial Wildlife and Forest Carbon Adaptation to assess the current forest stewardship plan in how it addresses climate impacts to wildlife, especially endangered, threatened, and declining species, their habitats, and the existing carbon pool and carbon sequestration rates (for more on NIACS courses see Chapter 4).

The main actions resulting from the climate adaptation planning process include using silviculture and prescribed burns to favor oak and hickory tree species and determining a way to address forest pests, especially in hemlock forests. Interestingly, these actions were similar for both wildlife and carbon. Using silviculture and prescribed burns to open the forest canopy is expected to favor oak and hickory trees, which are projected to be better adapted and more resilient to future climate impacts, thus increasing carbon sequestration rates and the amount of carbon stored over the long term. These same actions are expected to increase vegetation diversity, and therefore wildlife diversity, including a plethora of Lepidoptera species that rely on oaks and other shade-intolerant trees as host plants, and increase water availability for vernal pools and wetlands to prevent them from drying out as much during warmer, drier summers.

Actions to open the canopy to favor oak-hickory forests are already incorporated into the existing forest stewardship plan on SMWMA and have been implemented on a limited basis over the course of ten years. Monitoring the response of bird species to the treatment, as indicators of forest habitat and diversity, has been part of the plan. More recently, carbon estimates have been calculated and projected into the future based on proposed treatments and no treatment. Results of both indicate an increase in wildlife diversity, especially bird species of greatest conservation need (Figure 1), and a long-term increase in carbon sequestration rates post-treatment (Figure 2).
Figure 1. Average number of bird species of concern (orange bar) and all other bird species (blue bar) observed during breeding bird surveys on Sparta Mountain WMA. Pre-treatment surveys were conducted on site prior to treatment or conducted in 2008 and selected based on proximity to treatment sites within the same forest stand.

Figure 2. Projected carbon sequestration rates in 10-year increments under the proposed forestry activity (seed tree harvests) and no activity (no management). Projections were made based on inventories of Stand 18 on the SMWMA using the Forest Vegetation Simulator.

D) FINAL REFLECTIONS
Combining wildlife and carbon objectives in the climate adaptation planning allowed managers to determine actions that will benefit both and find compromise where actions/goals did not agree. For example, retaining hemlock using biocontrol methods would be beneficial for wildlife species dependent on hemlock forests, such as the Black-throated Green Warbler \((Setophaga virens)\), but that action is not recommended to defend existing carbon sinks. A compromise, therefore, could be to replace hemlock trees with surrogate coniferous tree species that will be better adapted to future climate conditions and not susceptible to Hemlock Woolly Adelgid; these surrogate trees could provide habitat for some, but not all, of the wildlife species currently supported by hemlocks. Another important takeaway is that focusing solely on maximizing carbon storage for climate change mitigation by eliminating active management from New Jersey forests may compromise important wildlife habitats and reduce opportunities for increasing long-term resilience of carbon sequestration and storage (Littlefield and D'Amato 2022).

III. CASE STUDY 3: REGIONAL SPONGY MOTH OUTBREAK AFTER DROUGHT

Written by H. Higgins, NE CASC

A) BACKGROUND

As discussed in Chapter 1, climate change is modifying weather patterns in the Northeast U.S. One example of such a change is the mid-2010s drought throughout much of the NEAFWA region, where such extended drought is atypical (Krakauer et al. 2019). Drought can directly stress trees, crops, and wildlife. Moreover, managing the interactive effects of atypical climate conditions is complex, as illustrated by the unexpected extreme outbreak of the Spongy Moth (formerly known as gypsy moth), \(Lymantria dispar\), an invasive forest pest.

Spongy Moth was first introduced to the U.S. via eastern Massachusetts in the late 19th century, escaped cultivation, and began feeding on native trees soon after (Liebhold et al. 1997). Since then, the species has cycled through periods of high population levels and widespread defoliation (millions of acres annually across the eastern U.S.), interspersed with many years of low population levels and only minor impacts. Oaks (\(Quercus\) spp.) are the primary host trees of Spongy Moths, but they also feed on others including aspens, willows, birches, and even hemlocks and pines when more favored hosts are depleted. They have an annual life cycle, and
their caterpillars, active in late spring and early summer, are the life stage responsible for feeding impacts on trees. Spongy Moth caterpillars pupate, become adults through the summer, and lay eggs in the autumn (Coleman et al., 2022). Therefore, effective Spongy Moth management practices are most effective when they align with the timing of the life stage they target.

During the 20th century, forest managers tested and developed techniques to manage the Spongy Moth population, some more successful than others (Elkinton & Roehrig, 2023). Chemical treatments, including growth inhibitors and mating disruptors, as well as more general insecticides, can be effective, but are not widely used in the region where this pest is already long-established (Elkinton & Roehrig, 2023). A native soil bacterium, *Bacillus thuringiensis* var. *kurstaki* (BtK), is effective at killing Spongy Moth and has been applied for large-scale aerial spraying, but can have negative impacts on related, non-target native insects, which provide important ecosystem services (Sample et al., 1996). Another biotic control now present in the ecosystem at large is a Spongy Moth-specific nucleopolyhedrosis virus (NPV), which is most effective at high population densities. Therefore, it often plays a role in decreasing large outbreak populations but does not appear to naturally curtail small or growing Spongy Moth populations.

However, the most surprising success of Spongy Moth control has been the fungus *Entomophaga maimaiga*, which was first released in the U.S. as a biological control of this pest in 1910 (Andreadis & Weseloh, 1990). Its impact was not noted until 1989, when it began killing large numbers of Spongy Moth caterpillars (Andreadis & Weseloh, 1990). Unlike NPV, its efficacy is not density-dependent, so it can be used as a population control even when Spongy Moth populations are relatively low. However, it is also dependent on optimal conditions for its own growth, specifically ample moisture early in the growing season. Its resurgence in the 1990s was linked to an abnormally wet year (Andreadis & Weseloh, 1990) and for roughly 25 years, it continued to keep Spongy Moth populations low, persisting in the environment through both natural spread and inoculative releases (Hajek et al., 2021).

**B) CLIMATE IMPACTS**

An uncharacteristic period of drought in the mid-2010s created unfavorable growing conditions for the biological control of *E. maimaiga*, particularly during the spring when both the fungus and caterpillar stage of the pest are most active. As a result, Spongy Moth populations increased drastically in several mid-Atlantic and New England states, defoliating 2.25 million
acres in 2017, the peak outbreak year (U.S. Forest Service, 2022). Most trees can recover from a single year of defoliation, but they can die after being subjected to multiple years of stress. Other factors also play a role in tree mortality, including water stress from the underlying drought and the compounding impacts of secondary pests, which exploit already-stressed trees (Coleman et al., 2022). It can be difficult to distinguish individual tree stress defoliation from mortality during the course of an outbreak, making the true estimation of impacts challenging. However, several states did experience substantial tree mortality after several years of Spongy Moth defoliation, particularly of oaks. This was the first major population boom of Spongy Moth, and subsequent defoliation event, since the 1980s. Since then, the outbreak has abated in some Northeast U.S. states, likely due to wetter conditions increasing *E. maimaiga* growth. However, the outbreak has begun, continued, or resurfaced elsewhere between 2020 and 2023.

Tree mortality has a wide range of ecosystem impacts, including on wildlife: mast-producing trees like oaks are an important food source for many species, and structural forest attributes associated with mature, living trees provide habitat for many others. In addition, tree leaf litter affects the nutrient balance of the forest floor, soils, and even nearby water quality. Mortality increases light availability on the forest floor, influencing regeneration, succession, and future species composition. Dead trees may also represent an economic loss to individual forest landowners and pose a hazard to nearby humans, requiring active management of safety risks. Finally, the loss of mature trees may reduce wellness benefits, particularly in residential and urban settings (Janowiak et al., 2021), and may also affect Indigenous lifeways and traditions associated with affected species (Daigle et al., 2019).

This complex chain of causes-and-effects is summarized in Figure 3, with an example timeline of their progression based on data from one state (Connecticut) in Figure 4.
Figure 3. An events chain showing several contributing causes, including climate-related ones, and some wildlife and habitat impacts of widespread tree mortality caused by recent Spongy Moth outbreaks in the Northeast U.S. Credit: Figure designed by Liz Sisk, U.S. Geological Survey, National Climate Adaptation Science Center
Figure 4. A timeline showing the progression of a key climate condition (springtime drought), biological response (defoliation by Spongy Moth), and ecosystem impact (tree mortality) in the Spongy Moth outbreak in Connecticut. Drought data are sourced from the National integrated Drought Information System, and Spongy Moth defoliation and mortality data are from the Connecticut Department of Energy and Environmental Protection. Credit: Figure designed by Liz Sisk, U.S. Geological Survey, National Climate Adaptation Science Center

C) MANAGEMENT ACTIONS

As described above, managers have used a variety of tools to control Spongy Moth and its impacts over the past century. However, their approach has shifted over time, and this section focuses on the management response to the mid-2010s and early-2020s outbreaks in the Northeast, including how climate adaptation has played a role in the approach. Insights and perspectives on this issue were derived from discussions with several staff members from different state agencies, supplemented by the information directly reported by states. These discussions focused largely on managing the habitat and wildlife impacts of Spongy Moth and less so on the insect pest itself. The process of managers reacting to a longtime threat re-emerging under novel conditions is summarized in the following sections and reflections on the successes, challenges, and lessons learned for future management applications.

i. Monitoring
Monitoring is one of the most-used and most useful tools for states managing Spongy Moth. Surveying and counting egg masses when they are laid in the autumn is a good indicator of Spongy Moth population levels for the following spring and summer, and states can prepare accordingly. For example, the Vermont Department of Forests, Parks & Recreation has been surveying Spongy Moth egg masses annually, utilizing nine established long-term monitoring plots, since 1984 (VT FPR). In 2019 and even more so in 2020 and 2021, the survey numbers increased, alerting the state to a population increase. Information from this monitoring program, along with an understanding of the fungus that usually keeps Spongy Moth populations in check, led state managers to decide against direct intervention, predicting that \textit{E. maimaiga} would resume its biological control activity before any major adverse outcomes, such as tree mortality, were realized.

“We do know that these are more short-lived outbreaks, typically, and furthermore, we know that one year of defoliation is not going to kill trees. Most times, two years of defoliation isn’t going to kill trees. So for that reason, I don’t think anybody viewed it as, ‘Oh my gosh, we must treat right now. Otherwise, everything’s going to die.’ Treatment is, of course, a last resort, and hopefully, we’ve weathered this, and hopefully, the outbreak is on its way out,” said one state employee. “Having broken out of abnormally dry and drought conditions, there was a small amount of crossing our fingers that there was going to be elevated fungal activity coupled with that increase in moisture. So we were hopeful, but also, the science backs it up that once you do have adequate moisture, it’s likely that the fungus is going to have a resurgence. And so that was […] helpful for us to say, OK, we don’t need to be heavy-handed with this approach and spray.”

Aerial surveys are another important monitoring tool for Spongy Moth and other forest health issues. Many states conduct surveys one or more times each year, flying over the state to observe and map areas of defoliation and other impacts. Based on the time of year, the visual signature of the impact, and in conjunction with ground-truthing, defoliation events and other disturbances captured in these surveys can be attributed to specific agents. States also have established forest monitoring plots, such as the federal Forest Inventory and Analysis (FIA) program. The informal reports and observations of state lands staff, who are on the ground and interacting with forested properties more regularly, are also valuable sources of information for states.
A limitation of monitoring is that the scale and frequency of these data can be difficult to translate to practical management efforts. For some continuous monitoring plots, because each individual plot is only visited once every several years and the plots are widely spaced, they may not be an adequate source of information on their own about Spongy Moth impacts on forests. Similarly, aerial surveys may be flown only once a year, so they may not capture the extent of impacts in a given year. One state employee explained, “Our foresters have a really good understanding of their districts. Bringing in other data sources and thinking at a strategic scale to prioritize actions are one of our biggest challenges.” In addition, Spongy Moth phenology is projected to shift due to climate change, specifically linked to increasing temperature, which may require adapting the timing of monitoring and other management efforts (Gray, 2004).

ii. **Public forest management**

Forestry holds critical insights for managing wildlife habitat and other benefits. To address recent Spongy Moth outbreaks, state forest managers have responded both directly, in targeted ways, and by incorporating climate and forest health concerns into their broader management objectives. Direct responses include salvage logging and cutting dead trees that may pose safety hazards, such as near roads. More broadly, the traits that help forests be more adapted to a pest like the Spongy Moth—such as species diversity and optimal stocking/density, achieved via strategic thinning—also serve other benefits, including resistance and resilience to other climate stressors. As one manager said, “There were a few projects that were spurred on by that awareness that, yeah, Spongy Moth is around, it’s a concern, but not really prioritizing that. We weren’t able to say, ‘OK, we’ve got a Spongy Moth vulnerability map that includes, how droughty are the soils? What’s the oak dominance? Is there already some kind of regeneration debt between an oak-hickory overstory and an oak-hickory understory? Is the repeated defoliation causing a change in these communities?’” Several state foresters used tools, including the Northern Institute of Applied Climate Science (NIACS) Adaptation Workbook and the associated Resistance-Resilience-Transition framework, to think about adaptive management strategies and how their current practices might already help achieve desired future conditions. A state project manager explained, “The NIACS framework for implementing adaptation strategies is core to all of our staff’s thinking.”
In reality, many forest and other habitat management practices are well suited to improve overall forest health, including weathering uncertain future conditions and new or resurging threats. As one forester put it, “Traditional, good, conservation-minded forestry is climate-smart forestry, too.” Other times, managers must consider conflicting goals and priorities. For example, species like red maple are considered “climate winners,” well-adapted to future abiotic conditions in the Northeast (Iverson et al., 2008) (and in the case of red maple, less susceptible to Spongy Moth), but they may be undesirable for other reasons—including less mast production and comparatively lower economic value than currently dominant species such as oaks (Southern New England Stumpage Price Report, 2021). Particularly when managing in response to a widespread mortality event like the one caused by Spongy Moth, promoting regeneration of desirable species is an important way to shape future forest composition.

Climate directly impacts typical harvesting practices: in the Northeast, many harvests take place in the winter, on frozen ground, both for operational ease and to minimize disturbance to the soil. However, warmer and shorter winters have shortened this optimal logging season, creating ripple effects on local wood systems (Contosta et al., 2019). These effects are particularly apparent when a disturbance such as Spongy Moth causes extensive mortality. “When the outbreak happens, you have the confluence of all the factors: Everyone’s busy, the existing loggers can’t be everywhere at once, the prices crash because there’s a glut of oak wood going to the market. So it’s a tough situation for landowners,” according to one forester. “The ability to respond in terms of management, including cutting or harvesting, was pretty limited. The wood industry is atrophied in southern New England. You can have the best silvicultural plans in the world, but if you don’t have good loggers to carry them out, or a market for the wood, your ability to do so is going to be limited.”

Public and political concerns also shape state-level management, including responses to invasive pests. In Massachusetts, a logging moratorium on state lands was in place from 2009-2012; another was enacted in 2023. In the period between, state forest management has still been restricted and continues to be subject to a lengthy approval process. These moratoriums stemmed from public activism around climate change, specifically carbon storage by forests. However, they may have unintended adverse impacts, such as trees in denser, unharvested forests being more susceptible to Spongy Moth and other pests. The restrictions also limit the potential management responses to pests such as climate change, including the opportunity to respond
nimbly when a threat such as Spongy Moth reemerges. For example, one forest management proposal in Wendell State Forest was submitted in 2016, and harvesting was not completed until nearly four years later. A state employee commented, “There’s definitely parts of [Spongy Moth] decision support tools or decision-making frameworks that they have developed over the years that aren’t applicable. Oftentimes those frameworks talk about trying to time thinnings between outbreaks, and we just don’t do that. Our foresters would love to be able to maintain stands at lower density, but there’s just not the resources to be able to implement that much thinning…Three-quarters of our oak-hickory stands are above full stocking, and that’s tens of thousands of acres.”

iii. Public outreach and management assistance

Most forestland in the U.S., especially in the Northeast, is privately owned (Butler et al., 2021). With this in mind, outreach and assistance to private landowners is crucial to any widespread forest health issue, such as Spongy Moth. Each state handling the recent Spongy Moth outbreak took action to communicate the growing outbreak to the public: forest health bulletins and extension publications, newspaper articles, radio interviews, and in-person outreach were all tools used to spread awareness and management advice to the public. Spongy Moth, in particular, is a pest with noticeable impacts when populations are high, including in urban areas (Coleman et al., 2022). High numbers of caterpillars and egg masses were seen by members of the public, and educating them about the problem and its impacts was an important facet of the outreach effort. On the other hand, private landowners sought advice on how to manage Spongy Moth on their own properties. State agencies provided practical advice for individual tree protection, monitoring of impacts, and more intensive management methods.

This two-pronged approach can also be applied to other climate-related habitat and wildlife issues. Harnessing the public interest in a problem raises awareness of the issue, while more targeted suggestions for an impacted group, such as forest landowners, can extend management efforts in private lands where states cannot intervene directly. To this end, similar education and outreach efforts are underway in relation to the introduced emerald ash borer across the Northeast US.

Outreach and collaboration with not only the public, but with other agencies and entities, can be both a challenging and a successful element of management. In Vermont, to prepare for
increased Spongy Moth populations as the egg mass surveys predicted, multiple state agencies collaborated to inform private landowners about the issue and smooth the path for approval of aerial spraying of BtK, should individual landowners choose that option for their forests.

Cross-state communication is another opportunity to share information through avenues such as the Forest Stewards Guild’s oak resiliency project. As one manager in the project explained, “There are a lot of different monitoring protocols that are used by one agency, or by a certain landowner, but they’re not really standardized across the region. So, it’s hard to compare monitoring data from Connecticut, Rhode Island, and adjacent parts of Massachusetts, for example…A great value of the Oak Resiliency Project was that it got landowners from the three different states working with each other, and I hope that we can continue that in some way.”

iv. Final reflections

In retrospect, state forest managers used a variety of tools at their disposal to address the Spongy Moth outbreak as it played out in the mid-2010s. They incorporated climate adaptation into their planning using tools like the NIACS Adaptation Workbook, monitored Spongy Moth populations and activity to track climate-induced changes, and coordinated management and educational efforts with private landowners and the general public. They also faced substantial challenges, including incomplete data availability, policy constraints on forest management activity, and related public opposition. For example, BtK spraying on public lands was not considered a viable management option for many states, for a combination of reasons that always included negative public perception. According to one land manager, “it was never really on the table.”

The mid-2010s Spongy Moth outbreak is still lingering in some parts of the Northeast, even interacting with different tree stressors. In Massachusetts, drought and Spongy Moth defoliation continued into the 2020s, albeit over smaller areas. A late spring frost in 2023 (Armanini and Sennott, 2023) caused tree mortality in areas that had been repeatedly defoliated by Spongy Moth, adding another element to the sequence of events in which Spongy Moth can be a contributing factor to tree mortality with other, climate-related ultimate causes. Climate-driven Spongy Moth outbreaks in the Northeast are not limited to the recent past—nor are they unique to this region. Elsewhere, climate and insect pests have interacted in a cycle of disturbance and amplification, such as in western North America where climate change caused a
native bark beetle’s shifting phenology range expansion, allowing it to kill millions of hectares of trees, which subsequently increased fuel loads for widespread wildfires (Grimm et al., 2013).

Anticipating future Spongy Moth outbreaks, “there’s more you can do to be a bit more proactive and less reactive,” said a forester. Recent climate modeling suggests drier conditions will continue to inhibit *E. maimaiga*, leading to increased Spongy Moth defoliation in the next 25 years (Liu et al., 2023). Conversely, another recent study analyzing past outbreak data found that climate change-driven temperature increases shifted tree species composition to less favorable Spongy Moth hosts, resulting in less defoliation (Haynes et al., 2022). Regardless, many forest management practices intended to limit the impact of Spongy Moth are consistent with broader adaptation strategies for climate change impacts. These forest management actions were implemented in the 2010s, and adaptive forest management practices will continue into the future, including those that prioritize wildlife habitat and other benefits.

**IV. CASE STUDY 4: WRACO DAM FAILURE AND FUTURE-ORIENTED INFRASTRUCTURE**

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**A) BACKGROUND**

In Michigan, like the Northeast, extreme precipitation events are becoming more intense and more common, and historical data are no longer an accurate standard for future conditions (*Cite chapter 1*). These changing baselines were actualized in April of 2014 when northern Michigan had an extreme rain event that caused a new water control structure to fail. This event demonstrated the importance of future-oriented planning and catalyzed a climate adaptation effort that both mitigated current risks and created a future-oriented plan. This example, detailed below, demonstrates how managers can respond to and plan for shifting baselines in a systematic way over a broad geography.

In 2012, the Michigan Department of Natural Resources Wildlife Division (DNR) purchased a shallow lake with wetlands habitat, and after the purchase they found a dam in poor repair on the lake. The DNR acted quickly to create a new, improved water control structure. The new water control structure was engineered to withstand a 100-year flood event, the standard for DNR water control structures. However, the risk of a 100-year flood event was based on average
precipitation during the past 30 years, and was not adapted to recent or future precipitation trends. This mismatch led to the water control structure failing within a few years when northern Michigan got a four-inch rain event over three days. Even though the water control structure failed, this event ended up as a near miss because there was no loss of life or property, but still led to tangible changes.

B) MANAGEMENT ACTIONS

A DNR technician was assigned to visit every water control structure that the DNR owned, determine what level of repair it was in, and determine if it was still meeting DNR goals (e.g., providing fish and wildlife habitat.) This assessment was needed not only to assess what state of repair dams were in (many were around 50 years old) but also because over the decades Beavers (*Castor canadensis*) had returned to the landscape and were creating marsh habitat naturally. These assessments led to systematically repairing water control structures that still provided valuable fish and wildlife habitat, and decommissioning those structures in areas where beavers were creating the same habitat in the surrounding landscape. Decommissioning old dams prevents a dam failure and subsequent flood events and allows Beavers, i.e., unpaid ecosystem engineers, to take over. Removing dams also cools the water temperature in streams, maintaining habitat for cold water fish and mitigating the impacts of climate change.

While removing dams has benefits, it can be controversial. Removal means draining lakes that private property owners value, and in some cases, property owners have fought the removal of failing dams. Some property owners want to keep dams because they create lake-front property and their sense of place and home value is tied to the lake. These lakes also support fishing, boating, and other recreation which some local communities want to maintain (House, 2023). This tension demonstrates that adaptation planning and modification may be ecologically and economically sound, but still face social blowback.

V. LITERATURE CITED


https://doi.org/10.1093/ee/nvab068


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