Chapter 9
Risk Assessment

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9.1 A Risk-Based Framework

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What is "risk" in the context of climate change? How can a "risk-based framework" help assess the effects of climate change and develop adaptation priorities? *Risk*

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can be described by the likelihood of an impact occurring and the magnitude of the consequences of the impact (Yohe 2010) (Fig. 9.1). High-magnitude impacts are always risky, even if their probability of occurring is low; low-magnitude impacts are not very risky, even if their probability of occurring is high. Applying this approach to forest management is challenging because both the likelihood of occurrence and the magnitude of the effects may be difficult to estimate (especially at local scales) and often depend on past and current land use, and the timing, frequency, duration, and intensity of multiple chronic and acute climate-related disturbances.

Despite these challenges, there is much that we do know and it is possible to begin thinking about how to develop a risk-based framework for evaluating the effects of climate change on forests. A risk management framework simply means that risks are identified and estimates are made for their probability of occurrence and their impact. Where we have sufficient knowledge, this framework provides a means to quantify what is known, identify where uncertainties exist, and help managers and decision makers develop strategies with better knowledge of risks.

Climate change will affect forest ecosystems, and the risk of negative consequences to forests and associated biosocial systems will probably increase (Ryan and Archer 2008). However, predicting these risks is difficult because of uncertainty in almost all aspects of the problem. How can we incorporate uncertainty into an analysis of risks and subsequent management decisions? Regional and local projections of climate change are uncertain (Baron et al. 2008; Joyce et al. 2008; Fagre et al. 2009). Despite these uncertainties, climate science has advanced to provide a set of robust climate change projections: the climate is warming, the probability of large precipitation events is increasing, seasonal patterns will be altered, and extreme events are more likely (Solomon et al. 2007). These tendencies

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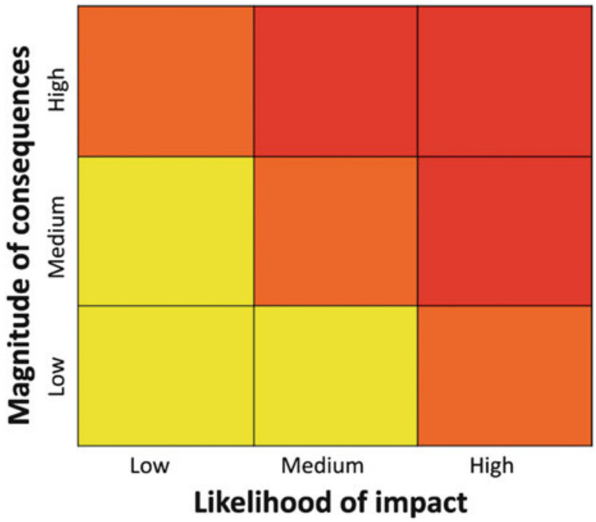

Fig. 9.1 A conceptual risk framework used to help identify risks associated with climate change and prioritize management decisions (Yohe and Leichenko 2010). Colors represent varying degrees of risk (*red* = highest, *yellow* = lowest). In a qualitative definition of consequence, low = climate change is unlikely to have a measurable effect on structure, function, or processes within a specified timeframe (e.g., 2030s, 2050s 2090s); medium = climate change will cause at least one measurable effect on structure, function, or processes within a specified timeframe; and high = climate change will cause multiple or irreversible effects on structure, function, or processes within a specified timeframe. In a qualitative definition of likelihood, low = climate change impacts are unlikely to be measurable within the specified timeframe, medium = climate change impacts are likely to be measurable within the specified timeframe, and high = climate change impacts are very likely (or have already been observed) within or before the specified timeframe

are becoming more apparent in observations across the United States and will affect forest resources nationwide (Karl et al. 2009).

A key challenge is to determine how climate change will alter local biosocial systems, trigger threshold-dependent events, and create nonlinear interactions across interconnected stressors on forest resources (Fagre et al. 2009; Allen et al. 2010), and further, how climate change effects can be addressed by local management actions. Forest managers have experience adapting forest management practices to climatic variability and disturbance regimes. For example, conifer plantations are often managed in short rotations, which limits exposure to risks from insects, wildfires, and windstorms. In mixed-age hardwood forests where management is often less intensive (e.g., where partial harvests are the norm), managers simultaneously choose trees to remove and trees in the understory to release for the next generation of growth. Hence, by using silvicultural techniques to select the species, density, and age class distribution of the next generation of forest, susceptibility to a range of future threats can be modified.

Given what we know about climate change, a robust decision-making approach is needed that acknowledges uncertainty, incorporates system vulnerabilities, and evaluates assets critical for making management decisions (Australian Government 2005; Baron et al. 2008; Joyce et al. 2008; Fagre et al. 2009; Ranger and Garbett-Shiels 2011). A risk management approach provides a framework for identifying management options for climate change, where uncertainties are recognized and management objectives and priorities are explicitly addressed (McInerney and Keller 2008; Yohe and Leichenko 2010; Dessai and Wilby 2011; Ranger and Garbett-Shiels 2011; Iverson et al. 2012). This approach incorporates vulnerability assessment, identifies priority actions relative to management goals, identifies critical information needs, and provides a vision of short- and long-term strategies to enhance the flexibility of management decisions and reduce the probability of poor decisions (Australian Government 2005; Peterson et al. 2011a). This approach also promotes a shift from reactive adaptation to proactive adaptation and coping management (Ranger and Garbett-Shiels 2011) (see Chap. 8), including the following general strategy:

- Identify actions to avoid, that is, avoid choices that lead to less flexibility to adjust to changing conditions.
- Implement "no regrets" management to cope with current stresses and increase resilience to anticipated climate-related stresses.
- Make decisions that integrate across landscapes and governance and that include all concerned and affected stakeholders.
- Develop activities that have strong links among observations, research, and management to understand how ecosystems and social systems are changing, help make decisions, understand thresholds, and help adjust future management and research.

The risk framework must consider the biosocial context of the system being evaluated, reflecting the contribution of forest ecosystem services to different communities and the capability of forest systems to withstand different climate stresses. Providing a more thorough consideration of sources of uncertainty allows for improved development of management strategies, which include key socioeconomic properties. This integrated and multi-sectoral approach will incorporate an improved assessment of risk and current management capacity, and will identify critical uncertainties that may exist under future scenarios if novel consequences emerge.

Case studies using a risk-based framework and concepts are discussed in the following sections on water, carbon, fire, forests, and birds. They are intended as examples, using different approaches to convey risk assessment, and will hopefully create interest by scientists and land managers in developing risk assessments for the effects of climate change on a wide range of forest resources.

9.2 Risk Case Studies

9.2.1 Water Resources

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The importance of forest watersheds for producing and maintaining high quality water flows is well accepted in the scientific literature (Barten et al. 2008). High quality flows are a function both high water quality (e.g., low nutrients and suspended sediment) and regulated flows (e.g., dampened extremes, stable base flows). Climate change will interact with and alter watershed processes in ways that may affect the ability of forests to maintain high quality water flow (Milly et al. 2008; Vörösmarty et al. 2010; see Chap. 3). Some of these interactions will be direct, for example, changes in total precipitation and extreme precipitation events that alter rainfall-runoff relationships. Others will be indirect, such as climate driven disturbances and changing forest species that can alter evapotranspiration and hydrologic flow paths. Altered precipitation and disturbance regimes will interact, for example, through the effects of a combination of extreme wildfires and more intense storms on water quality.

The strong dependency of humans and aquatic organisms on forest watersheds for drinking water (ecosystem services) and habitat (ecological flows), respectively, adds an inherently high level of risk to any climate-based changes in hydrologic processes (Milly et al. 2008; McDonald et al. 2011). For example, an increased frequency of low (or zero) flows could have severe impacts on aquatic species and municipal water supplies. Climatic, biophysical, socioeconomic, and demographic conditions differ greatly across the United States, so the vulnerability of forest-derived water resources (i.e., where water flows are insufficient to meet human needs or sustain aquatic ecosystems) to climate change is not uniform. Integrated water balance models have been used to identify vulnerable regions across the globe and in the United States and to evaluate how changes in climate and human demography will affect future vulnerabilities (Vörösmarty et al. 2010; USDA FS 2012).

In this case study, we develop a risk-based assessment approach based on the assumption that the ratio of precipitation (P) to potential forest evapotranspiration (PET) provides a simple index of water supply (Fig. 9.2). PET sets a theoretical upper limit on plant water use (transpiration) and evaporative losses and is driven by climatic factors (e.g., air temperature, solar radiation, wind speed, etc.). Because plant physiognomy affects water use, PET is usually referenced to specific vegetation types (e.g., forest vs. grass) (Allen et al. 1994). When P is greater than PET,

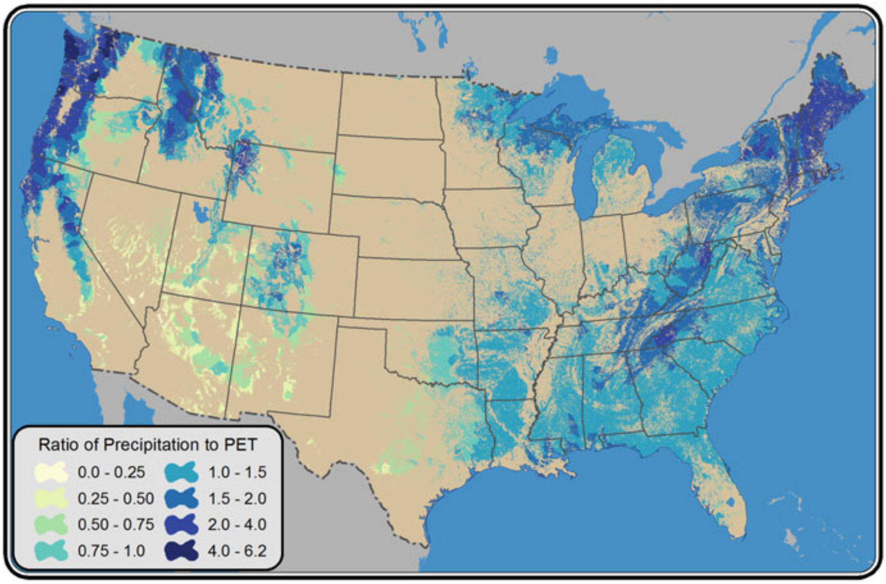

Fig. 9.2 Ratio of precipitation (P) to potential evapotranspiration (PET) for forests in the continental United States. PET was calculated using the Hamon (1961) model

excess water is available for streamflow and groundwater recharge. This excess water provides surface water and groundwater recharge for potential human use, and supports aquatic ecosystems.

Using a ratio of P and PET as an index of vulnerability, risk for both ecosystem services and ecological flows in areas where P/PET is less than 1 would increase if P decreases (Fig. 9.3). In contrast, areas where P/PET is considerably higher than 1 may be less vulnerable to lower P and higher PET, but could in some cases be more vulnerable if higher P is associated with more extreme rainfall and flooding. Vulnerability is also a function of the socioeconomic and ecological ability to rapidly mitigate or adapt to impacts. For municipal water supply, examples of socioeconomic responses include reduced demand through conservation, increased available water supply through more storage capacity, and redistribution via intra- and inter-basin transfers. For ecological flows, aquatic species will be especially vulnerable to changes in both annual flow and intra-annual flow because of limited capacity for mitigation and adaptation. Therefore, the negative consequences of reduced ecological flows on aquatic species would potentially be severe.

Using a risk-based framework in combination with the P/PET map for forests in the continental United States, we project that the Southwest has the highest risk for detrimental effects of lower precipitation and extended droughts. Some areas

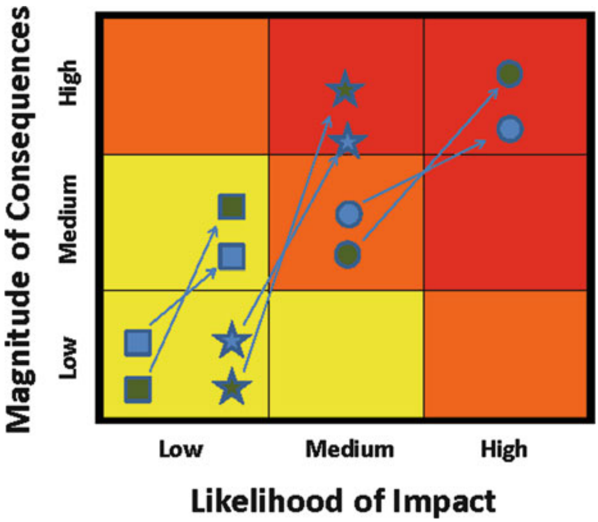

Fig. 9.3 Changes in risk (*arrows* indicate transition from current risk to future risk) as a consequence of increased drought frequency and severity. P/PET < 1 = ○; P/PET > 1 = □; P/PET = 1 = ☆; *green* = ecological flows, *blue* = ecosystem service flows. Risks are higher for ecological flows than for ecosystem service flows because some of the risks to the latter can be offset by engineering and conservation

in the Southwest already employ conservation, storage, and inter-basin transfers to meet current needs and offset current risks, although these measures are unlikely to be sufficient to offset the effects of climate change. In contrast, we project that areas in the upper Lake States, Northwest, and Northeast, where P/PPT is considerably higher than 1, have much lower risk from higher temperature and lower precipitation, because P is already in excess. However, these areas may have higher risk from extreme rainfall events that increase flood frequency and severity (e.g., Halofsky et al. 2011). In this case, responses may require re-examining current flood zones and riparian buffer widths, changing road and culvert designs to accommodate higher flows, enhancing storm water management, and changing designs for roads and infrastructure.

In areas where P/PET is near 1 (e.g., eastern portions of the southern United States), direct and indirect climatic changes can tip the P/PET balance in either direction (Jackson et al. 2009). If large deviations from a P/PET ratio near 1 have been historically infrequent, then neither aquatic organisms nor socioeconomic systems may have the capacity to withstand extreme events (droughts, heavy rainfall), and they may be at even greater risk to climate change than areas that have developed under frequently dry (P < PET) or frequently wet (P > PET) conditions.

9.2.2 A Framework for Assessing Climate Change Risks to Forest Carbon Stocks

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Forest ecosystems can reduce the effects of climate change through sequestration of carbon (C) (Pan et al. 2011) as well as contribute to net emissions through tree mortality, wildfires, and other disturbances (Kurz et al. 2008). A conceptual framework for assessing climate change risks to forest ecosystem C stocks facilitates efficient allocation of efforts to monitor and mitigate climate change effects. For example, the U.S. National Greenhouse Gas Inventory (NGHGI) of forest C stocks (Heath et al. 2011) can be used as a basis for developing a climate change risk framework for forest C stocks (Woodall et al. n.d.).

A risk framework for forest C stock incorporates consequence and likelihood as components of risk (Fig. 9.4; compare to Fig. 3.1). One of the most critical future consequences of climate change on forest C stocks is the shift from C sink (net annual sequestration) to C source (net annual emission). Although global forests currently sequester more C than they emit on an annual basis (Pan et al. 2011), it is unclear if or for how long this trend will continue in the future (Birdsey et al. 2006; Reich 2011). If the strength of the C sink decreases and forests became net emitters of C and other greenhouse gasses (GHG) (e.g., methane) a positive feedback loop may be created in which climate change effects may further exacerbate forest C emissions. Likelihood can be phrased as the probability of a C stock becoming a net emitter of C. Likelihoods would be minimal for individual C stocks that are least affected over short timespans (e.g., 50–100 years). Taken together, the C risk framework hinges on the concepts of a "status change" in which forest C stocks transition between C source or sink and a "tipping point" at which forest systems might collapse with concomitant emission of C and potential positive feedbacks that may exacerbate climate change.

We assert that the consequences of a C stock becoming a net emitter of C is directly related to its population estimate over a region of interest. In this case study, it is the C stocks of individual forest pools for the entire United States as reported to the Intergovernmental Panel on Climate Change to meet United Nations Framework Convention on Climate Change requirements (USEPA 2011a, b). If a pool is largest in the United States, then that pool has the largest consequence on global climate change if it is entirely emitted. All current U.S. forest C stocks represent nearly 25 years of U.S. GHG emissions at current emission rates (Woodall et al. 2011). The pools and estimates (Tg C) of C stocks in 2008 (Heath et al. 2011) are ordered as: soil organic C (17,136 Tg C), aboveground live biomass (16,854 Tg C), forest floor (4,925 Tg C), belowground biomass (3,348 Tg C), and dead wood (3,073 Tg C).

The likelihood of any individual C stock becoming a net emitter of C is an emerging area of research. For the purposes of this risk framework (Fig. 9.4), it

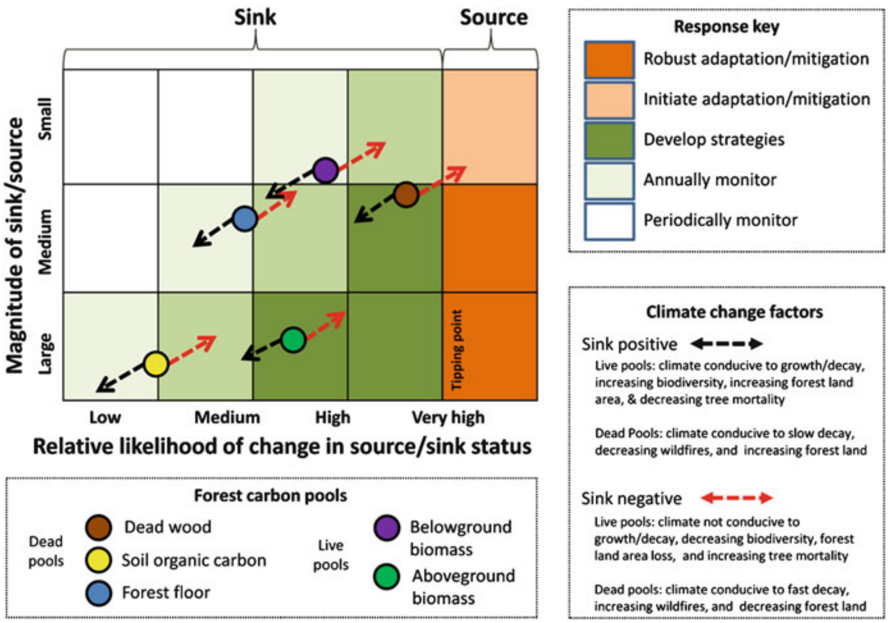

Fig. 9.4 Climate change risk matrix for forest ecosystem carbon (*C*) pools in the United States, in which climate change may cause C pools to move in a positive (sink = net annual sequestration) or negative (source = net annual emission) direction. Likelihood of change in C stocks is based on the coefficient of variation across the national Forest Inventory and Assessment plot network (x-axis). Size of C stocks is based on the U.S. National Greenhouse Gas Inventory (y-axis). Societal response (e.g., immediate adaptive response or periodic monitoring) to climate change events depends on the size and relative likelihood of change in stocks. The dead wood pool, a relatively small stock, exhibits increasingly high variability across the landscape and therefore may be affected by climate change and disturbance events such as wildfire. In contrast, the forest floor is a relatively small C stock, and has low variability. Potential climate change effects are not incorporated in the matrix, because they represent many complex feedbacks both between C stocks (e.g., live aboveground biomass transitioning to the dead wood pool) and the atmosphere (e.g., forest floor decay)

is proposed that the likelihood of a C stock becoming a net emitter is related to the empirical variation in the stock across the diverse ecosystems and climates of the United States. If climate change occurs such that a mesic boreal forest ecosystem becomes a xeric mixed-hardwood shrubland, then the contemporary range in variation in C stocks between those systems indicates likelihood of C emission. For example, if forest floor C stocks change minimally regardless of climate, then in turn climate change would least affect these stocks. As an initial appraisal of empirical variation in C stocks across the United States, the coefficients of variation (percentage) of individual plot-scale measurements of C stocks (Forest Inventory and Analysis; Heath et al. 2011) across the United States are ordered as dead wood (126.9 Tg C), belowground biomass (107.8 Tg C), aboveground live biomass (104.5 Tg C), forest floor (73.7 Tg C), and soil organic C (67.6 Tg C).

Although climate change events can alter natural variation in C stocks, when compared to contemporary levels, these estimates of variation provide a starting point for a risk framework.

When the consequences and likelihoods of forest C stocks becoming net emitters of C are viewed together, a cohesive approach to monitoring and managing risk emerges. Given the magnitude of potential emissions coupled with the natural variability in these stocks at the continental scale, annual monitoring of dead wood and aboveground live biomass C stocks are needed. In addition, strategies to mitigate negative climate change events (e.g., droughts) can be undertaken. The major research gap in such an approach is how far a pool would move within the risk framework after a climate-related event (the length and direction of the negative/positive arrows in Fig. 9.4). For example, if forest lands convert to grasslands as a result of reduced precipitation and lack of tree regeneration, how would the aboveground biomass pool align itself within the risk framework? Despite the qualitative nature and research gaps within the forest C stock risk framework, this approach provides a conceptual means of identifying priority research needs and a decision system for mitigating climate change.

9.2.3 Risk Assessment for Wildfire in the Western United States

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Wildfire is one of the two most significant disturbance agents (the other being insects) in forest ecosystems in the western United States, and in a warmer climate, will drive changes in forest composition, structure, and function (Dale et al. 2001; McKenzie et al. 2004). Although wildfire is highly stochastic in space and time, sufficient data exist to establish clear relationships between some fire characteristics and some climatic parameters. An assessment of wildfire risk in response to climate change requires brief definitions of the terms "fire hazard" and "fire risk," which are often confused in the scientific literature and other applications (Hardy 2005). *Fire hazard* is the potential for the structure, condition, and arrangement of a fuelbed to affect its flammability and energy release. *Fire risk* is the probability that a fire will ignite, spread, and potentially affect one or more resources valued by people. The most common means of expressing wildfire risk are (1) frequency, (2) a combination of intensity (energy release) and severity (effects on forests, structures, and other values), and (3) area burned.

Fire frequency, the number of fires for a particular location and period of time, differs by region as a function of both lightning and human ignitions, with the requirement that fuels are sufficiently dry and abundant to burn. Lightning ignitions dominate mountainous regions that have convective weather patterns (e.g., most of the Rocky Mountains), whereas human ignitions dominate regions with little lightning and high human populations (e.g., southern California). Modeling studies (+4.2 °C scenario) (Price and Rind 1994) and empirical studies (+1.0 °C scenario) (Reeve and Toumi 1999) suggest that lightning frequency will increase up to 40 % globally in a warmer climate. Although no evidence exists to suggest that recent climate change has caused an increase in lightning or fire frequency in the West, lightning may increase as the temperature continues to rise (Price and Rind 1994; Reeve and Toumi 1999). Assuming that human population will increase throughout the West, it is reasonable to infer that human ignitions will also increase in most regions. Even if the sources and numbers of potential ignitions do not change, a warmer climate may facilitate increased drying of fine surface fuels (less than 8 cm diameter) over a longer period (on a daily and seasonal basis) than currently exists (Littell and Gwozdz 2011), allowing more potential ignitions to become actual ignitions that will become wildfires.

Fire intensity, or energy released during active burning, is directly proportional to *fire severity* in most forests, and can be expressed as effects on vegetation, habitat, and in some cases, human infrastructure. Results of modeling based on a doubled carbon dioxide (CO_2) emission scenario suggest that fire intensity will increase significantly by 2070 in the northern Rocky Mountains, Great Basin, and Southwest (Brown et al. 2004). Fire severity and biomass consumption have increased in boreal forests of Alaska during the past 10 years (Turetsky et al. 2010), and large, intense fires have become more common in California (Miller et al. 2008) and the southwestern United States during the past 20 years. However, interannual and longer term variability in climate-fire relationships can affect trends, making it difficult to infer whether climate change is responsible. Longer time series of fire occurrence, when available, will allow better quantification of the influence of multidecadal modes of climatic variability (e.g., the Pacific Decadal Oscillation, Atlantic Multi-decadal Oscillation). Fire intensity and severity are a function of both climate and land use history, especially the effects of fire exclusion on elevated fuel loads, and forests with high fuel loading will continue to be susceptible to crown fire in the absence of active management (see below).

Fire area has a stronger relationship with climate in the western United States than does either fire frequency or severity/intensity. An empirical analysis of annual area burned (1916–2003) for federal lands in the West projected that, for a temperature increase of 1.6 °C, area burned will increase two to three times in most states (McKenzie et al. 2004). In contrast, a mechanistic model projected that, for the same temperature increase, area burned will increase by only 10 % in California (Lenihan et al. 2003). Using the 1977–2003 portion of the same data set used by McKenzie et al. (2004), Littell et al. (2009) stratified fire area data by Bailey's ecoprovinces (Bailey 1995) to account for fire-climate sensitivities. On average, the model explained 66 % of the variability in historical area burned by combinations of seasonal temperature, precipitation, and Palmer Drought Severity Index. In most forest ecosystems in the northern mountainous portions of the West, fire area was

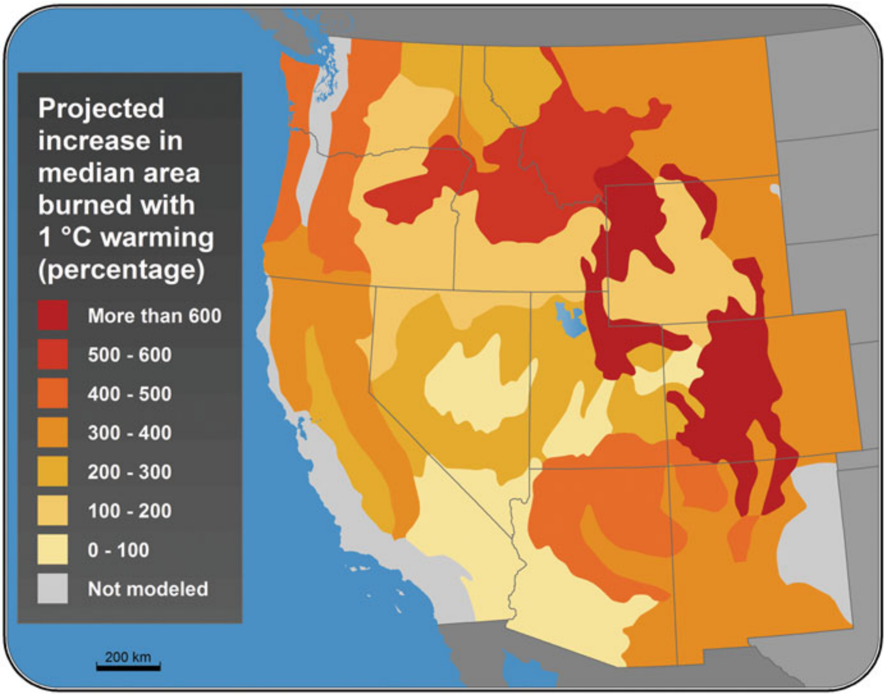

Fig. 9.5 Percentage of increase (relative to 1950–2003) in median area burned for western United States ecoprovinces for a 1 °C temperature increase. Color intensity is proportional to the magnitude of the projected increase in area burned (From Littell (n.d.))

primarily associated with drought conditions, specifically, increased temperature and decreased precipitation in the year of fire and seasons before the fire season. In contrast, in arid forests and woodlands in the Southwest, fire area was influenced primarily by the production of fuels in the year prior to fire and secondarily by drought in the year of the fire.

Littell (n.d.) projected the statistical models of Littell et al. (2009) forward for a 1 °C temperature increase, calculating median area burned and probabilities that annual fire area would exceed the maximum annual area burned in the historical record (1950–2003). Fire area is projected to increase significantly in most ecoprovinces (Fig. 9.5); probability of exceeding the historical maximum annual burn area varied greatly by ecoprovince (range 0–0.44). For the Northwest, the projected increases in area burned are consistent with those found by Rogers et al. (2011) using the MC1 simulation model. A weakness of the statistical models is that, if the projected increased area burned were sustained over several decades, then at some point the large areas burned and decreasing fuel loads would result in less area burned than projected by the models. Neither statistical nor process-based models can satisfactorily account for the effects of extreme fire years and biophysical thresholds that may be exceeded in a much warmer climate.

Based on information summarized above and on expert judgment of the authors, the effects of climate change on fire risk are summarized for fire regimes that occur in forests of the western United States (Table 9.1). We estimate risk for a 2 °C increase, which is more likely by mid-twenty-first century than the more conservative temperature scenarios used by McKenzie et al. (2004) and Littell et al. (n.d.). All fire regimes in forest ecosystems would experience some increase in fire risk. Low-severity and mixed-severity fire regimes dominate dry forest ecosystems of the West and would incur the greatest overall risk in terms of land area. High-severity regimes cover less land area, so they would have less influence on large-scale ecological changes; however, local effects could be significant, particularly where high-severity fire regimes occur close to large population centers, where socioeconomic exposure could be high even if probability of an event were low.

Management of fire risk is a standard component of fire management in the United States. Fire suppression has traditionally been used on both public and private lands to reduce fire area and fire severity. Increasing area burned will provide significant challenges for federal agencies and other organizations that fight fire because of the high cost of suppression and difficulty of deploying firefighters to multiple large fires that may burn concurrently and over a longer fire season. Fuel treatments in dry forest ecosystems of the West can greatly reduce the severity of wildfires (Johnson et al. 2011) (see Sect. 6.5), although funding is available to treat only a small percentage of the total area with elevated fuel loadings. Fuel treatments that include mechanical thinning and surface fuel removal are expensive, especially in the wildland-urban interface, and in a warmer climate, more fuel may need to be removed to attain the same level of reduction in fire severity as is achieved under current prescriptions (Peterson et al. 2011b). Allowing more wildfires to burn unsuppressed is one way to achieve resource benefits while reducing risk, although this approach is often politically unacceptable, especially when fire threatens human infrastructure and other values. Managing fire risk will be one of the biggest challenges for forest resource managers in the West during the next several decades.

9.2.4 Risk Assessment for Forest Habitats: Case Study in Northern Wisconsin

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We used a risk matrix to assess risk from climate change for multiple forest species by discussing an example that depicts a range of risk for three tree species in

Table 9.1 Likelihood and magnitude of increased wildfire risk for fire regimes in forests of the western United States, based on a temperature increase of 2 °C[a,b]

Risk parameter	Fire regime: Low severity	Moderate severity	High severity	Rationale for risk ratings
Frequency:				
Likelihood	Moderate	Moderate	Moderate	More fires will occur in all forests because of longer fire seasons and higher human population. In low-severity systems with low fuel loads, more fires will maintain resilience to fire and climate change; in low-severity systems with high fuel loads, more fires will cause more crown fires. In moderate-severity systems, more fires could convert them to low-severity systems. In high-severity systems, even a small increase in fire frequency will have a large effect on forest structure, function, and carbon dynamics.
Magnitude	Low	Moderate	High	
Overall risk and potential action	Low; no action recommended	Moderate; encourage fire prevention in high population areas	Moderate; encourage fire prevention in high population areas	
Intensity/severity:				
Likelihood	Moderate[c]	Moderate	Low	In low-severity and high-severity systems, fire intensity and severity will probably be higher because of more extreme fire weather and elevated fuel loads for the next few decades. In high-severity systems, fuel moisture, not quantity, is limiting, so intensity and severity will not change much; crown fires are always intense and kill much of the overstory.
Magnitude	Moderate[c]	Moderate	Low	
Overall risk and potential action	Moderate; increase fuel treatment area and fuel removal	Moderate; increase fuel treatment area and fuel removal	Low; no action recommended	
Area burned:				
Likelihood	High	High	Moderate	All fire regimes will experience more area burned. This will be especially prominent in drier, low-severity and moderate-severity systems. In high-severity systems, more area will burn, and although the percentage increase will be less than in other systems, it will have significant local ecological effects.
Magnitude	High	Moderate	Moderate	
Overall risk and potential action	High; greatly increase fuel treatment area, allow some fires to burn	Moderate; increase fuel treatment area, allow some fires to burn	Moderate; no action recommended	

[a]Risk ratings are qualitative estimates based on information summarized above and on expert judgment of the authors

[b]Fire regimes are defined as (1) low severity: 5- to 30-year frequency, less than 20 % overstory mortality (dry mixed conifer forests and woodlands); (2) mixed severity: 30- to 100-year frequency, patchy and variable overstory mortality (mesic mixed conifer and drier high-elevation forests); and (3) high severity: more than 100-year frequency, more than 80 % overstory mortality (low-elevation conifer and wetter subalpine forests)

[c]Fire intensity/severity are expected to increase in the next few decades, but they may decrease if fuel loadings are sufficiently reduced over time

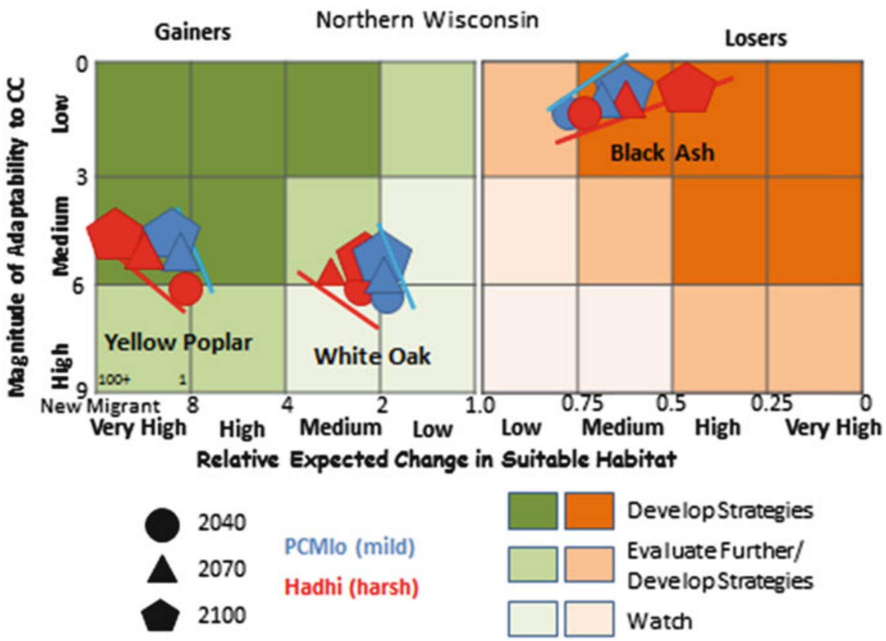

Fig. 9.6 Risk matrix of potential change in suitable habitat for three tree species in northern Wisconsin that are expected to either lose habitat (*black ash*), gain habitat (*white oak*), or become a potential new migrant because of newly appearing habitat (*yellow poplar*)

northern Wisconsin. We define risk as the product of the likelihood of an event occurring and the consequences or effects of that event. In the context of species habitats, likelihood is related to potential changes in suitable habitat at various times in the future. Consequences are related to the adaptability of a species to cope with the changes, especially the increasing intensity or frequency of future disturbance events. Data were generated from an atlas of climate change for tree species of the eastern United States (USDA FS 2011).

A risk matrix allows resource managers to determine which species need adaptation strategies, further evaluation, or monitoring programs. We adopted an established risk matrix structure (Yohe 2010; Yohe and Leichenko 2010; Iverson et al. 2012) to assess the likelihood of exposure and magnitude of vulnerability (or consequences) for three tree species in northern Wisconsin (Fig. 9.6). Much of the climate change literature focuses on potential decreases in forest species ("losers"), but increases may also pose management challenges, so the matrix was modified to include species or forest assemblages that are projected to increase in suitable habitat in the future ("gainers") (Fig. 9.6). The risk matrix is demonstrated for black ash (*Fraxinus nigra* Marsh.) (loser), white oak (*Quercus alba* L.) (gainer), and yellow poplar (*Lireodendron tulipifera* L.) (new migrant).

Black ash carries more risk because, among other disadvantageous traits, it has low resistance to emerald ash borer (*Agrilus planipennis* Fairmaire), which currently threatens all ash species in North America (Prasad et al. 2010). White oak is expected to gain habitat in northern Wisconsin, because it is well adapted to drier conditions and increased disturbance. Relative to other species, projected risk over time for this species is relatively low. Yellow poplar is not now recorded in northern Wisconsin, and as a potential new migrant into the region, it may provide new opportunities for habitat and wood products.

Using methods described in the DISTRIB system (Iverson et al. 2008, 2011; Prasad et al. 2009), data for the likelihood (x-axis) are based on a series of species distribution models to assess habitat suitability for 134 tree species in the eastern United States, for current and future (2040, 2070, and 2100) climatic conditions. "Likelihood" in this context is, for any point in time, the potential that a section of forest within a specified region will have suitable habitat for a given species relative to its current suitable habitat. In this example, we used 2 global change models and 2 emission scenarios (PCMlo and Hadhi) to elicit a range of possible risks, from low to high, associated with future climates. The matrix shows high variation between the modeled output, with Hadhi causing larger changes in suitable habitat for all species. For black ash, which loses habitat, the x-axis ranges from 0 (complete loss of habitat over time) to +1 (no change in habitat over time). For white oak, which gains habitat, the x-axis ranges from +1 to +8. For yellow poplar, a species entering new habitat, the range is confined to the leftmost column of the graph. These numbers themselves are not a direct scale of "likelihood," but rather are scales of future:current importance values.

Consequences in this context are related to the adaptability of a species or forest assemblage under climate change, based on a literature assessment of species biological traits and capacity to respond to disturbances that are likely to occur within the twenty-first century, including how those disturbances will be affected by climate change. Data for this axis come from a literature-based scoring system, called "modification factors," to capture species response to climate change (Matthews et al. 2011a). This approach was used to assess the capacity for each species to adapt to 12 disturbance types and to assess nine biological characteristics related to species adaptability. Each character was scored individually from −3 to +3 as an indication of the adaptability of the species to climate change. The mean, scaled values for biological and disturbance characteristics were each rescaled to 0–6 and combined as a hypotenuse of a right triangle; the resulting metric (ranging from 0 to 8.5) was used for the y-axis of the risk matrix (Fig. 9.6). Because several disturbances (e.g., floods, droughts, insect attacks) are expected to increase over time, we also used a formula based on modification factors to enhance relevance for certain factors from 2040 to 2100.

The risk matrix provides a visual tool for comparing species risks relative to changing habitats associated with climate change. Trajectories displayed in the matrix reveal insights about species response to climate change and can be considered in the development of potential adaptation strategies, although they cannot account for non-linear responses to extreme climate and altered disturbance regimes. The risk matrix can also help organize "climate change thinking" on a resource management team and communicate information to stakeholder groups and the general public. Finally, the risk matrix can be used to assess climate change risk for a variety of resource disciplines, and although the metrics may not be derived from the same methodologies, the capacity to rate one species against another, or one location against another, provides a consistent approach to managing climate change risk.

9.2.5 Risk Assessment for Bird Species: A Case Study in Northern Wisconsin

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Species distribution models for 147 bird species have been derived using climate, elevation, and distribution of current tree species as potential predictors (Matthews et al. 2011b). In this case study, a risk matrix was developed for two bird species (Fig. 9.7), with projected change in bird habitat (the x-axis) based on models of altered suitable habitat resulting from changing climate and tree species habitat. Risk was evaluated for three time steps (2040, 2070, 2100) and based on two climate models and two emissions scenarios (Hadhi, PCMlo).

To assess the y-axis of the matrix (Fig. 9.7), we used the System for Assessing Vulnerability of Species (SAVS) (Bagne et al. 2011; Davison et al. 2011) to estimate species adaptability to future changes, including disturbances. The SAVS tool is based on 22 traits that represent potential areas of vulnerability or resilience with respect to future climate change. Each trait forms the basis of a question that is scored according to predicted effect (reduced, neutral or increased population). By selecting responses for each question, a user creates a score that represents relative vulnerability to climate change effects, with higher positive values indicating higher vulnerability. Scores were calculated considering all 22 traits and divided among 4 categories: habitat, physiology, phenology, and biotic interactions. To calculate a baseline that could be used to compare current versus future vulnerability, we zeroed out individual questions for traits relating to exposure to future conditions and calculated a score based on the intrinsic characteristics of a species that reflect its sensitivity to population declines as a result of stochastic or other events.

Northern Wisconsin is near the edge of the distribution of the northern cardinal (*Cardinalis cardinalis* L.) and offers relatively limited habitat opportunities because of current winter climatic conditions. However, with projected increases in temperatures for northern Wisconsin, the habitat for the northern cardinal is

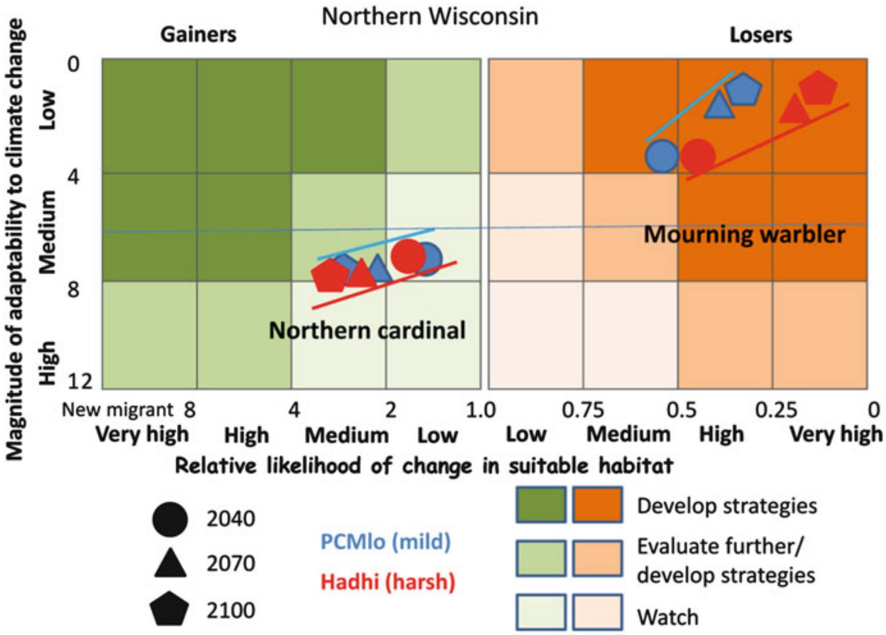

Fig. 9.7 Risk of the effects of climate change on the northern cardinal and mourning warbler, expressed as a combination of likelihood of habitat change (x-axis) and magnitude of adaptability (y-axis). Values are rescaled from calculations that used the approach in SAVS (Bagne et al. 2011; Davison et al. 2011)

projected to double by the end of the century (future:current habitat ratio of 2.2). The northern cardinal uses habitats ranging from shrublands to forests, has a broad diet, and has been shown to be positively associated within an urbanizing landscape (Rodewald and Shustack 2008). The SAVS baseline scores indicate less vulnerability (−0.91) and that the species does not show increased vulnerability risk under climate change (−1.82). Characteristics such as adaptability of nesting locations and flexibility in reproductive time contribute to the less vulnerable score.

In contrast, the mourning warbler (*Oporornis philadelphia* A. Wilson) shows higher risk based on its more specialist nature, specificity to breeding habitats, and Neotropical migration life history. These innate traits make the mourning warbler more susceptible under current conditions (SAVS +3.64) and is also considered at an increased risk of exposure to negative effects of climate change (+5.45). The mourning warbler is primarily a boreal species and despite its use of early successional habitats and a positive response to some human disturbances such as timber harvest (Hobson and Schieck 1999), its occurrence in northern Wisconsin declined over a recent 16-year interval (Howe and Roberts 2005). Moving beyond contemporary changes, its habitat is projected to decrease to one third of its current range by the end of the century (future:current ratio as low as 0.13 or 0.33, depending on climate model). These potential changes in habitat are attributed to higher temperatures and loss of boreal forest habitat (Iverson et al. 2008). In addition, the premontane and montane tropical life zones inhabited by the mourning warbler during winter are predicted to be highly sensitive to climatic affects (Enquist 2002). Therefore, when viewed together, the likelihood and magnitude of projected climate change suggest high risk for this species, and an increased opportunity for the northern cardinal, whose habitat will expand into northern Wisconsin.

The general approach used here can be applied to a wide range of species, using either quantitative information or qualitative logic. The empirical statistical models used here provide insights on the broad-scale determinants of species distributions, but with some limiting assumptions. Models derived from mechanistic relationships that explore processes regulating population dynamics also demonstrate the importance of local climatic conditions on avian populations (Rodenhouse 1992; Anders and Post 2006), but they are available for only a limited number of species. The detailed parameterizations of process models also have important assumptions and can be difficult to apply across a broad array of species. Thus, more refined inferences on how climate change may affect avian populations will require careful consideration of both empirical and mechanistic approaches to modeling species distributions, including the influence of ecological disturbances on habitat, and threshold values for minimum habitat quantity and quality.

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