# Ecological Dimensions of Nontimber Forest Product Harvest

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**H**UNDREDS OF PLANT AND FUNGI SPECIES (chapter 2) are harvested each year for nontimber forest products (NTFPs) because they play important roles in the cultural (chapter 4), social (chapter 5), and economic (chapter 6) lives of individuals and communities across the United States. Most of these species are harvested from natural populations, where they play important ecological roles, including providing food, cover, and habitat for diverse wildlife, including pollinators, and contribute to nutrient cycling, hydrological cycles, and erosion control. The continued use of NTFPs is contingent on each species ability to persist over the long term in their landscapes and with the qualities that people value. Species persistence depends in part on their ecological characteristics, the human and plant communities in which they grow, as well as the threats they face. This chapter reviews the effects of NTFP harvest on plant individuals, populations, communities, and ecosystems and then presents case studies of seven heavily harvested NTFPs in the United States. We conclude with a summary of key findings and recommendations.

## 3.1
## Effects of Nontimber Forest Product Harvest on Plant Individuals and Populations

NTFP harvest systems can have impacts at multiple ecological scales—from individuals to populations, communities, ecosystems, and landscapes. Sustainable harvest requires populations to persist long term and that the harvest does not negatively affect community and ecosystem functions at multiple ecological scales (Small et al. 2011, Ticktin and Shackleton 2011).

The most direct impact of NTFP harvest is on the vital rates of individuals harvested for their parts: their survival, growth, and reproduction. For example, fern leaves (fronds) are harvested throughout the United States for food, foliage, and crafts. Frond harvest removes photosynthetic material, reduces available nutrients for new growth, and can decrease significantly the size of new leaves and leaf growth over time (Aderkas and Green 1986, Bergeron and Lapointe 2001, Ticktin et al. 2006).

Substantial changes in some vital rates can lead to important demographic and genetic changes at the population level (see review by Ticktin 2004). The resilience or vulnerability of populations to harvest varies according to life history traits, ecological, management,

and social context of harvest (table 3.1) (Castle et al. 2014, Ticktin 2015). For plants, two important factors that influence a species' resilience or vulnerability to harvest are the part of the plant harvested and the life history traits of the plant. The latter includes patterns of growth, reproduction, pollination, and dispersal of seeds or spores, distribution, and habitat specificity (table 3.1). For example, long-lived perennials like trees and shrubs have reproductive parts (flowers, fruit, and seeds) that tend to have very high potential for sustainable harvest. Meta-analyses of demographic studies reveal long-term population growth rates of long-lived species are little affected by decreases in fecundity (Franco and Silvertown 2004). Demographic studies of trees and shrubs have shown that very high levels of fruit harvest tends to be sustainable at the population level (see review by Ticktin 2004, Emanuel et al. 2005, Sampaio and Maës dos Santos 2015, Ticktin et al. 2012). As such, the widespread wild-harvest of edible fruits, nuts, and berries—for example, more than 1.6 million pounds (>725,000 kg) on Forest Service and Bureau of Land Management lands per year (Alexander et al. 2011)—is not expected to lead to population decline. However, the effects of fruit harvest on other species, such as frugivores, remains poorly studied

In contrast, demographic rates of long-lived perennials tend to be highly sensitive to decreases in adult survival (Franco and Silvertown 2004). Therefore, harvest of adult individuals or increases in mortality of adults, such as some kinds of bark (figure 3.1), root, rhizome, and bulb harvesting, may have large negative impacts on long-term population persistence (Chamberlain et al. 2013, Schmidt and Ticktin 2012, Small et al. 2011). High levels of adult mortality are a concern for numerous species in the United States that are commercially traded. For example, many medicinal plants that are commercially traded are valued for their underground organs or for the whole plant (Alexander et al. 2011), and are therefore harvested whole. The International Union for Conservation of Nature (IUCN) Red List of Threatened Species lists 66 U.S. plant species for which overharvest represents at least one of the threats; more than 75 percent of these are orchids and cacti—long-lived perennials collected as whole plants for the horticulture trade.

Demographic models of some species heavily harvested for their underground organs, such as the medicinal plants American ginseng (*Panax quinquefolius* L.) and goldenseal (*Hydrastis canadensis* L.), and edible

**Table 3.1**—Factors affecting the potential for sustainable NTFP harvest. Arrows indicate that the characteristics listed in the columns for high and low potential for sustainable harvest represent two ends of a continuum. Adapted from Castle et al. 2014, Cunningham 2001, and Ticktin and Shackleton 2011.

| Category | Attributes | Potential for sustainable use | | |
|---|---|---|---|---|
| | | **High** | **Medium** | **Low** |
| **Sociopolitical** | NTFP uses | Single or non-competing uses; harvest of selected size-classes only | ⟶ | Multiple conflictive uses and harvest of different or all size-classes |
| | Local ecological knowledge (LEK) | Harvest is a historical activity, and highly detailed and sophisticated LEK systems exist and can be applied. | ⟶ | Harvest is a new activity and no LEK developed yet |
| | NTFP management | Highly tended wild or maintained populations; farmed or domesticated | ⟶ | Uncontrolled collection from wild; cultivation not viable; open-access resource |
| | Governance systems | Secure tenure, institutional arrangements fit with social-ecological system, effective monitoring and enforcement | ⟶ | Insecure tenure, institutional arrangements are misfits for system, no monitoring or enforcement |
| **Economic** | Seasonality of harvest | Short season with high abundance | ⟶ | Available all year round |
| | Substitutability | Many species can provide the same or similar product | ⟶ | Only one or few species offer the same product |

species such as ramps (*Allium trioccum* Aiton), have demonstrated they can sustain only low levels of harvest (Christensen and Gorchov 2010, Nantel et al. 1996, Rock et al. 2004, Van der Voort and McGraw 2006) consistent with research on NTFPs with similar life histories elsewhere (e.g., Ghimire et al. 2008). For some species, the number and size of populations has decreased over time, and overharvest is considered a contributing factor (Mulligan and Gorchov 2004).

Focused harvest of large adult plants can lead also to a decline in plant size over time. For example, McGraw (2001) used U.S. herbarium specimens to show that American ginseng plants of the same age had declined in size significantly since 1900 across most of its range where harvest pressure was greatest. A decrease in size over time has also been shown for species elsewhere (Law and Salick 2005). Few studies exist on the genetic implications of harvest, but decreasing population sizes have been shown to cause inbreeding depression (Mooney and McGraw 2007) and decreased genetic diversity (Cruse-Sanders and Hamrick 2004) in American ginseng.

The findings discussed in this section do not imply that harvest of underground organs or of whole plants is always unsustainable. Traits such as the ability to reproduce vegetatively can greatly increase the resilience to populations to harvest (table 3.1). For example, the roots of the long-lived perennial

echinacea (*Echinacea angustifolia* DC.) are harvested for the herbal product market. Data from both Kansas and Montana show that about 50 percent of the roots that remain in the ground following harvest resprout, and the plants grow back from 6 to 10 inches below the surface (Kindscher et al. 2008). Similarly, goldenseal's abundant rhizomes permit some to remain in the soil after harvest and allow some regrowth (Sanders and McGraw 2005a). Density-dependent responses to harvest—where reduced abundance due to harvest results in increased demographic rates—can increase the potential for sustainable harvest (Schmidt et al. 2011, Ticktin et al. 2002). In addition, management such as replanting roots or root parts can greatly increase the potential for sustainability.

### 3.1.1
### Management and Local Ecological Knowledge for Sustainable Harvest

The methods used to harvest NTFPs influence plant persistence despite sustained harvest (table 3.1). Sustainable management practices for NTFPs that are harvested from natural populations can include weeding competitors, clearing the overstory to increase light, and planting seeds, root, or rhizome pieces or seedlings to replace, expand, or extend wild NTFP populations. For example, when harvesting bluedicks (*Dichelostemma capitatum* (Benth.) Alph. Wood) in California, American

**Figure 3.1**—Mature slippery elm (*Ulmus rubra*) tree. The bark is harvested for its medicinal purposes. Forest Inventory and Analysis data indicates increased mortality of this species throughout its range, suggesting that harvesting is having negative impacts. (Photo courtesy of Rob Routledge, Sault College, Bugwood.org.)

Indians harvest the corms and replant the cormlets, spare plants, and harvest after seeding to ensure replenishment of seed (Anderson 2005, Anderson and Rowney 1999). Similarly, when American Indians in the Great Plains harvested the roots of prairie turnips or tipsin (*Pediomelum esculentum* (Pursh) Rydb.), they planted seeds from ripe seed heads back in the holes created when they dug the roots. Loss of this activity has led to the prairie turnip paradox: this plant is now uncommon throughout most of its range because people no longer harvest and eat the root (and plant the seeds) (Castle 2006). However, it should be noted that prairie turnips are still traditionally harvested on the Crow and Standing Rock (Lakota) reservations (Ruelle and Kassam 2013). Studies show the ecological impacts can vary significantly according to the seasonal timing of harvest, timing of harvest in the plant's life cycle, and the frequency, intensity, and methods and size of

harvest (e.g., Albrecht and McCarthy 2006, Sanders and McGraw 2005a, Van der Voort and McGraw 2006). For example, Nantel et al. (1996) showed that for both ramps and American ginseng, "choosy" harvesters who collect mostly larger plants, have a much greater negative impact on population viability than do "busy" harvesters, who collect the same number of plants, but from a broader range of sizes (see box 3.1).

The decisions that harvesters make about how to harvest, manage, or steward NTFP populations are influenced by many factors, including cultural, social, political, economic, and ecological context. However, one key factor is harvesters' understandings of the ecological system. Generations of observation, experimentation, and adaptation by local harvesters often lead to development of detailed traditional or local ecological knowledge (chapter 4; Hummel and Lake 2015), including highly sophisticated local management practices for maintaining culturally and economically important resources and landscapes (Anderson 2005, Berkes 2011, Price and Kindscher 2007, Turner et al. 2000). NTFP populations managed by knowledgeable harvesters may show high growth rates under high harvest pressure, while populations of the same species managed by others may decline under much lower levels of harvest (Price and Kindscher 2007, Schmidt and Ticktin 2012, Ticktin and Johns 2002). As has been demonstrated for American ginseng, with the appropriate management, there is potential even for species at high risk of overharvest to be managed sustainably (e.g., Van der Voort and McGraw 2006). This highlights the value of cooperative management agreements for NTFP, where different stakeholders can be involved in making management decisions.

### 3.1.2
### Spatiotemporal Variation in Nontimber Forest Product Population Dynamics

An important challenge in assessing harvest effects and designing plans for sustainable management of NTFPs is that effects vary over the landscape (Sinclair et al. 2005, Souther and McGraw 2011, Ticktin 2004) as well as over time, as climatic and other conditions vary (Schmidt et al. 2011). For example, available environmental resources (light, nutrients, water) influence long-term population growth rates. Populations of the Hawaiian vine, maile (*Alyxia stellata* (J.R. Forst. & G. Forst.) Roem. & Schult.)—widely harvested for its fragrant

**BOX 3.1**
CASE STUDY:
**American Ginseng Response
to Climate Change**

Despite wide latitudinal distribution, American ginseng (*Panax quinquefolius* L.) is sensitive to relatively small increases in mean temperature (Souther and McGraw 2011, 2014). Stochastic demographic models show that warming as little as 1.75 °F is sufficient to increase extirpation risk of an average ginseng population (Souther and McGraw 2014). This sensitivity to warming is attributed to the tendency of ginseng populations to adapt to local climatic conditions (Souther and McGraw 2011, 2014; Souther et al. 2012). Long-term demographic studies, simulations, and experimental manipulations indicate that climate change will negatively impact this species, especially in the presence of co-occurring stressors, such as harvest (McGraw et al. 2013; Souther and McGraw 2010, 2011, 2014; Souther et al. 2012). Rapid rate of climate change, coupled with habitat fragmentation, serves to reduce gene flow among populations and decreases the likelihood of adaptation to climate change (Davis et al. 2005, Etterson and Shaw 2001, Shaw and Etterson 2012). While rapid evolution is possible, adaptation to climate change for species like ginseng, which is characterized by slow-growth, long generation times, and low rates of reproduction, likely will be slow relative to fecund species that disperse seeds over long distances (Souther and McGraw 2014). Adaptation potential likely will be further reduced by decreased genetic variation resulting from harvest, as genetically based phenotypic variation is a requisite for evolution to occur.

Recently, several species of thrushes, including the wood thrush (*Hylocichla mustelina*), hermit thrush (*Catharus gutatus*), and Swainson's thrush (*Catharus ustulatus*), have been identified as potential dispersers of ginseng seed (Hruska et al. 2014). Juvenile thrushes may disperse seeds up to 300 m from parental plants. However, such long-distance dispersal events are considered rare, as most thrush dispersal events result in seed movement of less than 100 m from the maternal plant (Smith et al. 2004). Nonetheless, the occurrence of long-distance dispersal, however infrequent, indicates a possible mechanism for ginseng to track climatic change. Long-distance dispersal may introduce warm-adapted, southerly or lowland genotypes into northern and upland populations, thus potentially increasing the likelihood of adaptation to novel climatic conditions (Hampe and Petit 2005). Experiments testing the adaption of American ginseng and other NTFPs to rapidly changing climate are needed.

stems—grow significantly faster in closed-canopy than in more open-canopy forests (Wong and Ticktin 2014). Root yields of wild-harvested osha (*Ligusticum porteri* J.M. Coult. & Rose) are higher in open meadows than in forested areas (Kindscher et al. 2013). Similarly, international research has shown that population growth rates and levels of sustainable harvest can vary significantly between habitats (Ghimire et al. 2008), and environmental gradients (e.g., drier versus wetter environments) (Gaoue and Ticktin 2008, Gaoue et al. 2011). In some places, NTFPs are more often harvested along roads than from the forest interior. Differences across space and time in the surrounding ecological community and in management add to the large spatiotemporal variation observed for many NTFPs.

### 3.1.3
### Community Interactions and Ecosystem Dynamics

Long-term sustainability of harvests depends upon numerous factors that operate across multiple scales and levels of ecological organization, requiring understanding of processes that go beyond individual populations and species (table 3.1). These include the maintenance of key interspecific interactions, resistance to herbivore pressure, resilience to natural and anthropogenic disturbances, appropriate landscape management, as well as competing land uses or management objectives.

A decline or loss of key interspecific interactions across tropic levels, such as pollinators, mycorrhizal fungi, and animal seed dispersers, may threaten the viability of NTFP populations and make them less resilient to harvest (table 3.1). For example, the pinyon jay (*Gymnorhinus cyanocephalus*) is a critically important seed disperser of pinyon pines (e.g., *Pinus edulis* Engelm., *Pinus monophylla* Torr. & Frém.) in western United States and has experienced a mean population decline of 4.3 percent annually from 1966 to 2012 (Chambers et al. 1999, Sauer et al. 2014). Continued decline of this bird species may threaten the long-term ability of pinyon pine to respond to increased climatic variability through loss of its long-range dispersal mechanism. Similarly, in the tropics, hunting of seed dispersers can have adverse effects on NTFP species, as has been observed for African crabwood (*Carapa* spp.), where subsistence hunting frequently targets the rodents that disperse its seeds (Forget and Jansen 2007).

Changes in population dynamics of species in the surrounding community, such as increases in populations of herbivores and/or the introduction of invasive species, can also threaten NTFP species and increase the potential for unsustainable harvesting (table 3.1). Overabundant deer and elk populations increase herbivory rates in many parts of the United States, threatening sustainable NTFP harvests of certain species in addition to impacting numerous other ecosystem properties (Cote et al. 2004). One study of white-tailed deer (*Odocoileus virginianus*) impacts on 26 species of NTFP trees and shrubs in a Nebraska forest-agricultural mosaic found that woody NTFP species with dense branching (e.g., redosier dogwood, *Cornus sericea* L.; weeping forsythia, *Forsythia suspensa* (Thumb) Vahl; white willow, *Salix alba* L.; purpleosier willow, *Salix purpurea* L.) were damaged the least by deer antler rubbing, while species with one or few unprotected stems (e.g., American black elderberry, *Sambucus canadensis*; smooth sumac, *Rhus glabra*) were damaged the most (Hygnstrom et al. 2009). Levels of browsing damage also varied by NTFP species with the most damage found in *Cornus* spp. and Chinese chestnut, *Castanea mollisima*. Similarly, the ability of American ginseng to withstand harvest is reduced with increasing populations of white-tailed deer (Farrington et al. 2009, McGraw and Furedi 2005).

The ability of ginseng to withstand harvest is also threatened by the invasive species, garlic mustard (*Alliaria petiolata*) (Wixted and McGraw 2009). In Hawai'i, many NTFP species gathered for cultural practices such as hula (chapter 4) are declining in population due to competition from invasive species. In cases like these, harvest may be a contributing factor to population decline (Ticktin et al. 2006), although not necessarily the main cause. Studies elsewhere have shown that factors like grazing and invasive species may make much greater contributions to the decline of NTFP populations than harvest (Endress et al. 2004, Mandle and Ticktin 2012, Ticktin et al. 2012).

Responses of NTFP species to natural and anthropogenic disturbances are trait-dependent (Mouillot et al. 2013) and specific to the particular life history adaptations of a given species. In general, species will most likely be adapted to a characteristic disturbance regime. Therefore, populations will persist when ecosystem dynamics are maintained within a natural (or historic) range of variability, although management toward natural variability is complicated by processes such as climatic variability or invasive species (Landres et al. 1999). Traits that enable populations to survive or increase following fire include resprouting, seed dormancy, fire-stimulated seed production, and fast growth to rapidly colonize large areas of mineral soil. Within the fungal genus *Morchella* (morel mushrooms), certain species fruit in forests that are more frequently burned while other species favor unburned forests. However, productivity (count and fresh weight) at the genus level is generally greater in burned forests than in insect-damaged forests, and is least in undisturbed forests (Pilz et al. 2004). Fire frequency and severity can also play a key role in NTFP response. Blueberry (lowbush, *Vaccinium angustifolium* Aiton and velvetleaf huckleberry, *V. myrtilloides* Michx.) production in eastern Canada is favored by low-intensity fires with limited burn depth (i.e., consumption of soil organic layers) (Duchesne and Wetzel 2004). Fire was used traditionally by native peoples to increase the production and quality of roots and berries, improve produce materials for basketry and other arts and technologies (Kimmerer and Lake 2001, Wray and Anderson 2003). Some NTFPs adapted to particular fire regimes, including fires set by American Indians, are now less common, including Bradshaw's lomatium (*Lomatium bradshawii*), huckleberries (*Vaccinium* spp.), camas (*Camassia* spp.), tobacco (*Nicotiana* spp.), and deergrass (*Muhlenbergia rigens* (Benth.) Hitchc.) (Anderson 1996, Boyd 1999). Fire frequency plays a key role in sustainability of NTFPs harvested by native people who lived in Yosemite Valley (Anderson 2005) and who live elsewhere (Schmidt and Ticktin 2012, Sinha and Brault 2005). NTFPs that benefit from increased light availability but are intolerant of surface fire may be favored by wind disturbance or biotic disturbances such as forest insects (e.g., bark beetles) and pathogens.

Many North American NTFPs are understory species and strongly influenced by effects of disturbance and forest management on characteristics of the forest overstory (chapter 2). Shade-tolerant and shade-adapted NTFP species are favored by levels of overstory canopy cover that differ from those preferred by shade-intolerant species. Heavy thinning of Douglas-fir (*Pseudotsuga menziesii*) forests in the Oregon Cascade Range reduced production of shade-adapted chanterelle (*Cantharellus* spp.) mushrooms (Pilz et al. 2006). Beargrass (*Xerophyllum tenax* (Pursh) Nutt.) produces more commercially valued leaves in moderate shade and canopy density compared to heavy shade (Higgins et al. 2004). Leaves with qualities prized by tribal weavers are

associated with sites with tree basal area less than 200 square feet per acre and densities less than 130 trees per acre (Hummel and Lake 2015). Studies elsewhere have shown that timber harvesting can have strong negative effects on the availability of NTFP species (Shanley et al. 2002; see also chapter 2, section 2.2). On the other hand, population growth rates and the potential for sustainable harvest may be greater in secondary forests than in old-growth forests for some NTFP species, such as terrestrial and epiphytic bromeliads (del Castillo et al. 2013, Ticktin and Nantel 2004).

Sustainable NTFP harvest also can be threatened where dominant land uses are competing and not complementary (table 3.1). Habitat transformation or destruction may be the main cause of decline for some NTFP species. For example, in the Great Plains habitat destruction (the plowing of more prairie lands for additional crop production in the last two decades) has had more negative impacts on coneflower (*Echinacea angustifolia*) stands than harvesting has (Kindscher 2006). Similarly, management of pinyon-juniper woodlands in the Great Basin region of the Western United States has arguably reduced the availability of the pine nut resource. In Florida, prime quality habitat for saw palmetto (*Serenoa repens* (W. Bartram) J.K. Small), which is heavily harvested for its medicinal fruit, appears to have decreased by one-half since 1945 (Mitchell 2014).

## 3.2
## Implications of Nontimber Forest Product Harvest for Ecological Systems

The effects of NTFP harvest on maintaining biodiversity, ecological interactions, and ecosystem functions have not received sufficient study, as the great majority of studies of ecological effects have focused on sustainability of harvest at the population level (Ticktin 2004). Forest farming (chapter 2), a land use approach for more profitable NTFPs such as ginseng, goldenseal, (Burkhart and Jacobson 2009) and black cohosh (*Actaea racemosa*) (Small et al. 2014), may have some adverse effects on forest ecosystems but may be less detrimental on ecosystem services than converting forests to agricultural land (e.g., Trauernicht and Ticktin 2005). Wild harvesting of NTFPs can have variable impacts on nutrient cycling and soil erosion, depending on plant parts harvested (reviewed in Ticktin 2004).

NTFP harvest may also influence community successional dynamics. In the Oregon Coast Range, modern strip harvesting methods for commercial moss production influence the ecological succession of epiphytic bryophyte communities and reduce species and age-class diversity, in contrast to historical effects of more patchy, less efficient moss production that likely increased species diversity (Peck and Frelich 2008). Incidental taking of bryophyte species of low commercial value is also common during commercial moss harvesting (Studlar and Peck 2007), and likely has adverse effects on community structure and biodiversity.

Although sustainability of NTFP harvest at the population level may be greater when harvested plant parts are reproductive (flowers, fruits, seeds) rather than vegetative (leaves, roots, apical shoots; table 3.1), harvest of forest fruits may have important impacts on ecological community dynamics and biodiversity. There is little published information on such effects, although a few studies in the neotropics have found negative effects of palm fruit harvesting on avian frugivore diversity and abundance (e.g., Galetti and Aleixo 1998) (Moegenburg and Levey 2002). Forest plantings of palms can increase populations of fruit-eating birds, but also change the composition of avian community toward fruit eaters (Moegenburg and Levey 2002). Tree nuts can be especially important resources for bird and mammal species (e.g., whitebark pine nuts and grizzly bears) (Mattson et al. 1991). Competition between human harvesters of pine nuts and other animals that rely upon these food sources has been little studied. There appears to be competition between insect and vertebrate herbivores for pinyon pine seeds, such that birds and mammals were able to increase their seed harvest when insect cone herbivores were experimentally excluded (Christensen and Whitham 1993). Such observations suggest that pine nut supply may be limiting for herbivore species and that increased human harvest of pine nuts may compete with requirements of other species for this important food resource.

Fruit harvest by people does not always imply conflict with other frugivores. For example, in Florida hundreds of tons of saw palmetto drupes, commonly called berries, are harvested annually (AHPA 2012). Most saw palmetto habitat, however, does not coincide spatially with those areas most important for black bear foraging within their primary ranges. Harvesting therefore does not appear to represent a critical threat to the food resources for black

bears outside of their primary ranges (Mitchell 2014). Saw palmetto derives its name from the "saw-like teeth of its petiole margins" and grows into thick, dense clusters and shrubs (Bennett and Hicklin 1998). These characteristics lead harvesters to concentrate their efforts on the berries easily reached from the outside of the thicket, leaving much of the fruit within it available to wildlife (Bennett and Hicklin 1998, Mitchell 2014). Unlike saw palmetto that has defenses such as thorns, American ginseng is an understory plant readily grazed upon by deer (Farrington et al. 2009, McGraw and Furedi 2005). Competition between harvesters and wildlife would be expected to be greater for these types of NTFPs.

## 3.3
## Sustainable Harvest of Key Nontimber Forest Product Species

Descriptions of seven key NTFPs harvested across the United States provide insight on the range of environmental and harvest contexts, responses to harvest, and research and management needs for NTFPs. For each of the seven key NTFPs, we discuss the responses to harvest in the context of the plant's ecology, natural history, and distribution. These species represent a sample of some of the heavily harvested NTFPs in the United States and were selected because they span a range of plant parts harvested and of ecosystems, and because, in contrast to most NTFPs, there has been a fair amount of research conducted on the ecological impacts of harvest.

### 3.3.1
### American Ginseng

American ginseng is an herbaceous understory species found in the eastern deciduous forest of North America (Anderson and Fralish 1993). Harvest of ginseng for the medicinal plant trade began over 200 years ago in Canada (Robbins 2000). During the period of peak harvest in the 1800s, the United States exported hundreds of thousands of pounds of dried ginseng root to Asia annually (Kauffman 2009, McGraw et al. 2013). Intensive harvest, like that which occurred during the 19th century, likely reduced overall ginseng abundance, mean population size, and genetic diversity (Case et al. 2007; Cruse-Sanders 2005; Cruse-Sanders and Hamrick 2004; Kauffman 2009; McGraw et al. 2003, 2013). Reduction in ginseng population size decreases per capita reproductive rates (Hackney 1999, Hackney and McGraw 2001) and increases the

risk of inbreeding depression (Mooney and McGraw 2007). Loss of genetic variation may inhibit ginseng's ability to adapt if environmental conditions change. Through such impacts, past harvest events continue to influence population performance today.

In 1975, ginseng was listed in appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (Robbins 2000). Since that time, the U.S. Fish and Wildlife Service must issue a "no detriment" finding for export to continue. While ginseng harvest programs vary among U.S. States, generally, regulations establish a harvest season and a minimum age/size limit for harvest. Harvester behavior, and regulations that govern harvest behavior, dramatically influence plant population growth (Van der Voort and McGraw 2006). In model simulations, population growth rates of harvested populations were similar to those of nonharvested populations *if* harvesters complied with regulations, *as well as* actively stewarded populations by planting seeds and limiting harvest intensity (Van der Voort and McGraw 2006). Conversely, harvest that occurred prior to annual seed set and that removed undersized plants resulted in precipitous population decline (Van der Voort and McGraw 2006). Given this, illegal and/or irresponsible harvest poses a significant threat to ginseng persistence and continued trade of the species (McGraw et al. 2010).

In addition to irresponsible harvest (McGraw et al. 2013, Van der Voort and McGraw 2006), deer browse, (Farrington et al. 2009, McGraw and Furedi 2005), increasing abundance of invasive species (Wixted and McGraw 2009) and climatic variability (Souther and McGraw 2010, 2011, 2014; Souther et al. 2012) negatively impact ginseng population growth. In combination, these stressors may influence viability in complex and unanticipated ways. For instance, harvest in combination with climate change increases extinction risk well above the additive effects of either factor alone (Souther and McGraw 2014). Because shifting environmental conditions exacerbate negative impacts of harvest, increased protection, particularly in the form of stricter enforcement of current harvest regulations, may be required to prevent extinction as climate shifts away from historic norms (Souther and McGraw 2014). Future research should focus on developing climate change-integrated harvest regulations and conservation strategies.

### 3.3.2
### Common Beargrass

Common beargrass grows in western North America in maritime and continental areas and over a variety of habitat and soil types (Hummel et al. 2012). It can be a significant component of subalpine meadows and dry, mixed-coniferous forest types (Higgins et al. 2004). Vegetative reproduction occurs in spring and summer (Peter and Shebitz 2006). Flowering—which varies with soil temperature, aspect, canopy cover, and elevation— has been reported as early as April (Vance et al. 2004), and as late as September (Maule 1959), and is most prevalent in the open or in canopy gaps (Maule 1959, Vance et al. 2004). When environmental conditions do not favor flowering or pollination a population can reproduce vegetatively (Vance et al. 2004).

Beargrass is difficult to cultivate, so most harvest occurs from natural populations (Hummel et al. 2012). The maritime distribution of the plant—from the mountains of northwestern Washington and southward into west-central California—is the main location for commercial and traditional harvesting. While the industry mainly exports the plant for use as decoration (Blatner and Alexander 1998, Hansis 1998) and has since the 1990s, for many American Indian tribes beargrass is a key fiber in traditional weaving and has been for centuries (Hummel and Lake 2015).

The preferred leaf properties of beargrass differ between commercial and traditional harvesters and thus, so do the preferred site conditions where harvest occurs. Commercial-quality beargrass leaves are deep green, long, wide, and absent any discoloration (Schlosser et al. 1992, Schlosser and Blatner 1997). These leaf properties seldom exist under open forest canopy (Higgins et al. 2004). In contrast, tribal harvesters prefer a mid-green color, regardless of leaf length or width and weaving style (Hummel and Lake 2015). Good sites for traditional leaf harvest have lower densities of large diameter trees and lower levels of down wood than do poor sites (Hummel and Lake 2015).

A key challenge to sustaining populations of beargrass on accessible sites and with desirable leaf properties is an apparent tension between forest conditions that are good for commercial versus traditional harvest (a dense versus open canopy, respectively). A related challenge is designing, implementing, and enforcing a system for monitoring beargrass harvest levels according to type.

Commercial harvest on government lands is monitored through a permit system, with tracking on permitted harvest volumes and not actual harvests. A system that allowed for monitored volumes by harvester group could be beneficial. The effects of different harvest methods and intensities are not well studied or documented. Future research on the following is needed: effects of harvest practices (different intensities, and resident versus migrant harvest methods); plant responses to different drivers (climate, shade, fire, light levels/ competition from overstory trees); phenology (growth, development, response to warmer drier temperatures projected for western North America are not well understood); persistence and viability of beargrass seed; competition-density relations (e.g., effects on whitebark pine seedling regeneration and reproduction, drivers and thresholds for flowering versus vegetative reproduction across longitudinal and latitudinal gradients).

### 3.3.3
### Black Cohosh

Black cohosh, a native Appalachian forest herb, has been extensively harvested for its commercial value as an herbal medicine used to address symptoms of menopause (figure 3.2). Black cohosh has been listed as one of the top 10 selling herbal supplements each year since 2002. The American Herbal Products Association (2007, 2012) estimates that between 1997 and 2010, more than 3.76 million pounds of black cohosh roots and rhizomes were harvested from the deciduous forests of the Eastern United States. The mean annual harvest from natural populations over the 14 years was approximately 268,000 pounds. As nearly all the black cohosh sold commercially is wild-harvested, the potential for harvesting impacts is considerable without proper management practices in place. Mitigating potential harvest impacts requires determining harvest intensities that have minimal impact and allow for post-harvest recovery and long-term population persistence (Small et al. 2011). After 3 years of intensive harvesting (66 percent of stems within a plot harvested per year), Small et al. (2011) measured a significant reduction in foliage area and stem production, relative to control plots. After 1 year of recovery, plots from which plants were subjected to intensive harvesting showed no improvement. Instead, most growth measures continued to diverge relative to control plots. Thus, the harvest of two-thirds of a population appears unsustainable. Sustainable management of black cohosh, and other medicinal plants, requires effective

**Figure 3.2**—More than ¼ million pounds of black cohosh (*Actaea racemosa*) roots are harvested from natural populations each year to supply the herbal medicine market. Long-term collaborative research with university partners indicate that populations are not recuperating from the harvest. *In situ* inventory of the roots is one the major challenges being addressed to support management efforts. (Photo credit: James Chamberlain, U.S. Department of Agriculture, Forest Service.)

and reliable inventory of marketable plant components. Chamberlain et al. (2013) used data from long-term sustainable harvest studies to develop a predictive model to estimate marketable rhizome biomass at the stand level. Slow-growing perennials that require extended periods to reach reproductive maturity are particularly vulnerable to over-harvesting. With increasing demand for medicinal forest plants, such as black cohosh, assessing harvesting practices and determining viable management approaches is critical to the long-term sustainability of natural populations of forest herbs (Elliot et al. 2014, Small et al. 2014, Vickers et al. 2015).

### 3.3.4
### Echinacea

Echinacea is a native perennial to the Great Plains of the United States and southern Canada, and its roots are harvested from wild populations (typically the top 10 to 15 cm of root) using shovels or other implements (Kindscher et al. 2008). Roots are used extensively as a medicine for treating illness, wounds, snakebite, rabies, and other ailments by at least 16 American Indian tribes (Kindscher 2016a, 2016b). It is an important immune stimulant and flu and cold remedy in the herbal products trade with significant international exports to Europe. Other echinacea species have been harvested from the wild from central and eastern United States, but the vast majority of roots harvested are *Echinacea angustifolia* (Riggs and Kindscher 2016). Other species, especially those already rare or uncommon, or in locations where their populations are limited, are at risk because they can be sold as echinacea roots.

Over-harvest of wild echinacea species' root has been a significant concern to the herbal product industry and conservationists. However, research has shown that harvest can be sustainable. In one study of commercial harvest in Kansas and Montana, 50 percent of harvested root reserves had resprouted after 2 years despite droughty weather (Kindscher et al. 2008). The length

of root harvested affected resprouting ability in that study, with those plants having less root length removed more likely to resprout. Because the plants can produce substantial amounts of seed annually, recovery from intensive harvest is possible if periods of nonharvest occur. Additional work has demonstrated that there is a greater than 120-year history of harvest in locations of mixed grass prairie in central Kansas that, based on research on harvest practices, ecological monitoring, and demographic modeling, also appears to be sustainable (Price 1999, Price and Kindscher 2016). Concerns about the persistence of echinacea populations remain, however, due to a variety of factors related to the conversion of prairie lands to croplands, herbicide application to control "weeds" in rangeland, extensive oil and gas development (including national grasslands in North Dakota), and other development. Wild echinacea harvest is sustainable but populations should be monitored due to persistent habitat threats and to a changing climate.

### 3.3.5
### Goldenseal

Goldenseal is an herbaceous understory species native to the central Appalachian Mountains, west to Missouri and Arkansas, and north into southern Ontario, and is most abundant within the Ohio River watershed. While quantified accounts of historic (pre-1900) abundance are not available, narrative accounts suggest this species has experienced marked declines in both number and size of populations. This decline is not well understood and several contributing factors are likely. Evidence suggests harvest pressure, which has increased over the past 10 to 15 years (Inoue et al. 2013), as well as habitat loss and herbivory by increased populations of white-tailed deer have contributed to goldenseal decline (Mulligan and Gorchov 2004).

While harvest of natural populations is undoubtedly a contributing factor to population decline, the species' abundant rhizomes promote that some vegetative propagules are left in the soil following harvest, and permit a degree of regrowth (Sanders and McGraw 2005a). For example, after a wild-harvest in WV in which only four stems remained, 932 stems were present the following year (Van der Voort et al. 2003). Stem densities then declined initially, before remaining relatively stable 2 and 4 years post-harvest. Similarly, ramet densities increased 210 percent from 2 to 4 years post-harvest in Ohio (Albrecht and McCarthy 2006). Since preharvest

densities were unknown in both locations, it is not known how long it takes populations to recover from harvest.

Decline of goldenseal also may be linked to its conservative growth rate; it is a slow-growing species with little reproduction from seed. Several studies have documented that population growth rates are at or just above replacement value (Christensen and Gorchov 2010, Sinclair and Catling 2005). Common garden experiments indicate that soil moisture is important for successful germination and growth, but deep litter layers may hamper growth (Albrecht and McCarthy 2009). Goldenseal persists for the entire first season of growth with only its two seed leaves (cotyledons); it does not produce true leaves until the second growing season. In this first year state, seedlings may not be able to emerge through thick litter or even dense herbaceous cover.

As population decline continues, gene flow between populations will likely decrease. More than 80 percent of the genetic variation is within populations (Inoue et al. 2013), suggesting limited gene flow between populations. However, plasticity within populations has been documented (Sanders and McGraw 2005b), which would allow some degree of response to varying site conditions. Most studies of goldenseal have been limited to 4 years or less. A thorough understanding of population recovery, microsite response, demography, and plasticity can only be reached from longer-term research. To elucidate goldenseal's ability to respond to environmental fluctuations and changing environmental and climatic conditions, these four topics should be prioritized with well-designed, longer-term (at least 10-year) studies. Population recovery approaches should be multi-tiered and focus on (1) limiting harvest, (2) maintaining genetic diversity within populations, and (3) promoting gene flow among populations. Accomplishing this first point may involve legislation in Canada and the United States, as well as multiple States. Accomplishing the second two points may require development of breeding and restoration programs.

### 3.3.6
### Osha

Osha, bear root, or *chuchupate* is a slow-growing member of the parsley family (Apiaceae), found in the southern Rocky Mountains in the United States and the Sierra Madre of Mexico. Across its range it is primarily found from 1830 to 3570 m in moist meadows, parklands, riparian areas, and forests. It has a long

history of use by American Indians and Hispanics (Latino/as) as its pungent and distinctively spicy roots are used to treat influenza, bronchitis, and sore throat, and a variety of other ailments, particularly those relating to the lungs and heart and general illness (Kindscher et al. 2013). Today osha roots are primarily wild-harvested for personal use, but some are sold to herbal product companies, a demand that has been increasing.

Osha is a large plant (up to 6 ½ feet tall) and apparently long-lived, with large root crowns and rhizomes, often weighing more than a pound per plant. Information available on the sustainability of osha harvest indicates that plants and populations can recover from some harvest pressure (Kindscher et al. 2017, Mooney et al. 2015). Population densities of geographically separated dense populations in the Rio Grande and San Juan National Forests average of 7.8 percent osha cover (Kindscher et al. 2013). In a manipulative field experiment in the Rio Grande National Forest (Colorado) established to analyze osha's recovery from harvest, a meadow site had 15 percent more mature plants and 58 percent more root mass than the adjacent forested site. This meadow population exhibited 10-percent cover (a dense stand) and had on average 1.8 oz dried root weight per 10.76 ft$^2$ (465 pounds/acre), while a forest population exhibiting a 9 percent cover had on average 0.48 oz dried root weight per 10.76 ft$^2$ (122 pounds/acre) (Kindscher et al. 2013, 2017). This plant is very productive, but long-term concerns, especially related to climatic variability, may have negative impacts on future populations, but more research is needed on this topic.

### 3.3.7
### Pinyon Pine

Approximately 11 species of North American pinyon pine produce edible nuts. The most important are Colorado pinyon (*P. edulis*), dominant throughout 56 million acres of pinyon-juniper woodlands of New Mexico, Arizona, eastern Utah, and Colorado, and singleleaf pinyon pine (*P. monophylla*), which is abundant throughout the Great Basin areas of Nevada and western Utah. Pine nuts were a staple food of native peoples in the Western United States for millennia (Simms 2008). Pine nuts remain culturally important to traditional gatherers, although they are harvested locally and in relatively small quantities (Mitchell and Roberts 1999). There is much potential for further development of the pine nut industry in the United States. The domestic market is estimated at $100 million,

where more than 80 percent of pine nuts consumed are imported (Sharashkin and Gold 2004). Pine nuts are valued as a gourmet food item used primarily in pesto, salads, and various Mediterranean dishes, and can be pressed to make oil and ground to make meal. Pinyon pine nuts compare favorably with other tree nuts in their nutritional content of protein, carbohydrate, and unsaturated fats, with *P. edulis* the richer in oils and *P. monophylla* in carbohydrates (Lanner 1981). Harvest of *P. monophylla* nuts is usually by collection of closed cones in early autumn, following which cones are heated or allowed to dry so that seeds fall out after the cone scales open. Harvest of the smaller-coned *P. edulis* is usually by collection of individual seeds on the ground (Little 1993). Good pine nut crops are produced every 4 to 7 years (Barger and Ffolliott 1972), although the frequency of good mast years can be greatly reduced during drought periods, and during high late-summer temperatures, by as much as 40 percent (Redmond et al. 2012). Redmond et al. (2012) observed a 40-percent reduction in cone production from 1974 to 2008 in *P. edulis* populations of New Mexico, associated with high late-summer temperatures at the time of cone initiation.

Ecological sustainability issues associated with pinyon pine nut harvesting include competition with wildlife species that rely upon the nuts, including pinyon jay (*Gymnorhinus cyanocephalus*), Clark's nutcracker (*Nucifraga columbiana*), and several species of seed-caching rodents (Vander Wall 1997), as well as possible adverse effects of nut harvesting on long-term population viability of the tree species. These aspects have been but little studied. An ecologically based, silvicultural management of pinyon woodlands for pine nut production has been advocated over many decades (Lanner 1993, Little 1941) but little progress has been made toward this goal, which will require field trials, adaptive management, and an integration of scientific and traditional ecological knowledge. Pine nut silviculture would likely include uneven-age management of nut pines on favorable sites (Gottfried and Severson 1993). Judicious pruning and thinning could be implemented to reduce fire risk, stimulate perennial herbaceous understory development, and encourage nut production. Thinning has been observed to stimulate cone production in other nut pine species, such as stone pine (*P. pinea*) stands in southern Europe (Moreno-Fernandez et al. 2013). Fundamental challenges include limits in silvicultural knowledge

(stand dynamics, tree spacing, pest control); high variability of good crop years; climate change effects; labor and transportation costs; and competing landscape management objectives that currently emphasize removal of large acreages of pinyon-juniper woodland for fire risk reduction and to create or maintain sagebrush-dominated ecosystems (e.g., Davies et al. 2011).

Further research is needed concerning the optimal spacing for nut production, pinyon pine stand dynamics in response to uneven-aged silvicultural management (Gottfried 2004), and potential impacts of pine nut harvesting on woodland biodiversity including closely associated species such as pinyon jay. The pinyon pine nut resource faces an uncertain future due to current management efforts aimed at maximizing sagebrush habitat, which often incorporate large-scale tree removals, as well as potential climate change effects. Increasing temperatures and altered precipitation regimes have already affected pinyon pine populations through reduced growth (Williams et al. 2013), increases in mortality (e.g., Clifford et al. 2008) composition, and ecosystem services such as carbon sequestration. Our analyses of longitudinal

data from unmanaged old forests in the Western United States showed that background (noncatastrophic, altered patterns of seedling recruitment (Redmond and Barger 2013), and declining cone production (Redmond et al. 2012). More research is needed to determine effects of climate change on cone production across environmental stress gradients and for different species, populations, and genotypes of pinyon pine.

### 3.3.8
### Saw Palmetto

The saw palmetto palm is considered to be the most common native palm in the United States, found from Georgia to Mississippi and throughout Florida (Bennett and Hicklin 1998), where it is considered most abundant (figure 3.3). Once thought a weed and pest by farmers and ranchers (Bennett and Hicklin 1998), recent research has focused on the plant's range, distribution, preferred habitats, interactions with other plants and wildlife, adaptation to fire, longevity (some palms are believed to be over 1,000 years old) and more (cf., Abrahamson 1995, Abrahamson and Abrahamson 2009, Carrington et al. 2000, Carrington and Mullahey 2006, Maehr et al. 2001, Takahashi et al. 2011, Tanner and Mullahey 2012).



**Figure 3.3**—Saw palmetto (*Serenoa repens*) berries are harvested, predominantly from pine forests of Florida. The habitat for this important medicinal forest product is at great risk due to potential rise in sea level. (Photo credit: Karan A. Rawlins, University of Georgia, Bugwood.org.)

Saw palmetto has a long history of use by American Indians in Florida as food, fiber, medicine, and more (Bennett and Hicklin 1998). The commercial harvest of saw palmetto drupes, or berries, for herbal remedies in Florida goes back at least to the early 1900s (Bennett and Hicklin 1998). Today demand for the berries derives both from the European pharmaceutical industry, which processes the berries into a standardized oil used to treat benign prostratic hyperplasia (BPH) symptoms, and from the growing demand from the U.S. herbal supplement market, where in 2012 it was the third best-selling herb in the United States (Lindstrom et al. 2013). Berries are harvested primarily from Florida, where they are dried and processed for industry (Bennett and Hicklin 1998, Mitchell 2014). Reported harvests between 1997 and 2010 ranged from 763 tons to 2,893 tons of dried berries (AHPA 2012), reflecting both variable availability and demand. Berry harvesters are often seasonally unemployed agricultural workers who harvest informally for cash to bridge that employment gap. Companies formally lease land in order to maintain access to berries, but an informal market for the berries exists in Florida (Bennett and Hicklin 1998, Mitchell 2014).

There has been speculation that harvesting berries within black bear ranges has led to recent conflicts between humans and bears, though there is no evidence of a shortage of berries within their ranges, or that a lack of food led to human-bear conflicts. Recent research illustrates (1) much saw palmetto habitat exists outside of current black bear ranges, and (2) saw palmetto habitat within bear ranges should be more than adequate to meet bear foraging needs so bears would not have to leave their ranges to find this favored food (Mitchell 2014). Bennett and Hicklin (1998) note that harvester collection activities usually center on accessible public lands and in areas close to buyers. It is theoretically possible that this might result in localized scarcity during years of low fruit production (Bennett and Hicklin 1998). Much more research needs to be done to understand the relationship between the saw palmetto harvest and frugivores, such as black bears, toward improved management of both.

The act of harvesting berries should not harm the palm as only clusters of the berries are harvested (cf. Bennett and Hicklin 1998, Mitchell 2014). To determine whether the harvest is sustainable we would need to know more about the habitats where it is found and the quality of these. We also need to know more about where people harvest and national and international demand. We need to better understand harvest rates and patterns and other potential drivers on fruit production, plant regeneration, and long-term population growth rates and viability. The extent of the informal market is unknown as is the amount of annual production that flows into the herbal supplement industry. Florida's ongoing population growth has made it the third most populous state in the Nation. This in turn drives development into natural areas, reducing saw palmetto populations. This, along with expected climate change effects, such as sea level rise, will result in a decrease of saw palmetto habitat, which could increase pressure on remaining populations. More research is needed in these areas.

## 3.4
## Effects of Climate Change on Nontimber Forest Products

NTFPs include a diverse array of species that span broad taxonomic and environmental boundaries. This makes assessing the vulnerability of NTFPs to climate change a challenge. However, the life history traits that distinguish the majority of NTFPs from other well-studied species (e.g., timber species) can provide insight of their potential demographic, evolutionary, and spatial responses to climate change.

### 3.4.1
### Potential and Observed Species Responses to Climate Change

Mean global temperature has increased approximately 1.2 °F over the last century, with over half of warming (0.7 °F) occurring in the last 3 decades (IPCC 2013). To persist as climate changes, species must respond plastically, spatially, or evolutionarily to novel climate conditions (Davis et al. 2005, Jump and Penuelas 2005, Parmesan 2006). Long-term studies and biotic resurvey projects show that certain species have responded to contemporary climate change in a manner consistent with expectations (Badeck et al. 2004; Hoffmann and Sgrò 2011; Parmesan 2006; Parmesan et al. 2000, 2013; Parmesan and Yohe 2003; Walther 2010). For instance, many species have shifted distribution northward or upward and advanced the timing of critical life history events, such as spring emergence in plants or migration in avian species (Badeck et al. 2004; Hoffmann and Sgrò 2011; Parmesan 2006; Parmesan et al. 2000, 2013; Parmesan and Yohe 2003; Pinsky et al. 2013; Walther 2010). Despite this

clear ecological fingerprint of climate change, there have been ecological surprises as well. A significant proportion of species, 20 percent or more depending on the datasets analyzed, either remain unchanged or respond opposite to expectations (Tingley et al. 2012, Wolkovich et al. 2012). In general, species at greatest risk of extinction are characterized by narrow distribution, small population size, and/or limited dispersal, or that occupy habitats at distributional limits, like arctic and alpine systems (Brook et al. 2008, Cahill et al. 2013, Franco et al. 2006, Lewis 2006, Sax et al. 2013, Slatyer et al. 2013, Thomas et al. 2004, Wilson et al. 2005).

### 3.4.2
### Vulnerability of Nontimber Forest Products to Climate Change

Range breadth is frequently used as a primary indicator of vulnerability to climate change driven extinction, because a narrow distribution may indicate sensitivity to changing climate as well as habitat specificity, which could preclude successful colonization of northerly or upland sites (Bellard et al. 2012, Brook et al. 2008, Thomas et al. 2004). At first glance, NTFPs—generally not characterized by narrow range—may appear robust to changing climate. However, specialization to local climate conditions may narrow the thermal niche of a species, thus increasing vulnerability. Such specialization, termed local adaptation or ecotypic differentiation, is common among plant species (Linhart and Grant 1996). In essence, locally adapted species evolve traits that optimize performance at mean, site-specific climatic conditions. While this specialization enhances local competitive success, directional change away from historic norms reduces fitness (Atkins and Travis 2010, Davis et al. 2005, Etterson and Shaw 2001, Kawecki and Ebert 2004, Linhart and Grant 1996). In this way, climate change may threaten widespread species that are seemingly robust to changing conditions, in an analogous way to those that are narrow-range endemics (Atkins and Travis 2010).

Species comprised of many, locally adapted populations are not only more vulnerable to climate change driven decline, and they also may be less likely to successfully track home-site conditions as climate changes (Davis et al. 2005). Local adaptation is promoted when gene flow among populations is low (Kawecki and Ebert 2004). Hence, species for which the formation of ecotypes is the norm are often characterized by restricted dispersal,

and thus exhibit low potential for rapid distributional shifts in response to changing climate. Many NTFP species display relatively limited dispersal distances, which increase the likelihood of local adaptation (Bennington and McGraw 1995, Gregor 1946, McGraw 1985), and hence climate change vulnerability (Davis et al. 2005, Etterson 2004). Climate change may interact synergistically with other stressors like harvest pressure to increase extinction risk (Brook et al. 2008, Mandle and Ticktin 2012, Souther and McGraw 2014).

### 3.4.3
### Climate-change Integrated Management of Nontimber Forest Product Species

Climate-change integrated management of ecosystems encompasses a wide range of strategies. Such strategies include the removal or reduction of stressors (e.g., invasive species) to increase ecosystem resilience to changing climate, conservation of habitat corridors to allow species to track climatic niches upward and poleward, conservation of climate refugia, and human-assisted relocation of species that are unable to shift ranges (Hannah et al. 2002, Keppel et al. 2012, Richardson et al. 2009). Effective habitat management and conservation of habitat corridors and refugia seemingly have few drawbacks and are believed to benefit a wide range of species within ecosystems (Gilbert-Norton et al. 2010). However, sedentary or stationary species, with low migration potential (e.g., characterized by short dispersal distances, low rates of seed production) may be unable to shift distribution in response to changing climate even if habitat corridors are preserved. Such traits characterize a number of NTFP species.

When species imperiled by climate change are unable to track shifting climatic conditions, persistence may be contingent on human assistance (Etterson and Shaw 2001, Jump and Penuelas 2005). Due to their economic and cultural value, and likely vulnerability to climate change-driven extinction, NTFPs are potential candidates for *ex situ* conservation programs, such as managed relocation. Managed relocation, also known as assisted dispersal, migration, or colonization, refers to the transport of species, populations, or propagules to higher latitudes or elevations as climate changes (Richardson et al. 2009, Schwartz and Martin 2013). Interpreted broadly, managed relocation also may include the intentional introduction of genes preadapted to future climate conditions

into a population's gene pool in order to accelerate adaptation to changing climate. Conceptually, managed relocation decreases extinction risk by assisting the colonization of an area with a climatic regime similar to which the species (or population) is adapted.

Because species may be adapted to other site-specific variables in addition to climate, managed relocation has the potential to negatively impact fitness when associations with key environmental factors are broken. Additionally, the introduction of nonlocal genotypes into a locally adapted population may cause outbreeding depression or lowered fitness of subsequent generations due to infiltration of nonlocal, and hence maladapted, genotypes (Frankham 1995, Kramer and Havens 2009, Pertoldi et al. 2007). However, gene flow from populations adapted to warmer climates may provide genetic variation and traits necessary to adapt to novel climatic conditions (Hampe and Petit 2005). Given the uncertain efficacy of managed relocation, considerable research is needed to determine the relative benefits and risks of this conservation strategy for NTFPs, as well as to develop methodological considerations to perform successful relocations (Benito-Garzón et al. 2013, McDonald-Madden et al. 2011). If managed relocation is considered a viable strategy, citizen-based managed relocation programs that enlist harvesters to intentionally disperse seeds to cooler habitats should be evaluated as a potential approach. Additional lines of inquiry should address the possible role of cultivation in NTFP conservation. Cultivation programs for many NTFPs exist and could serve as sources for managed relocation, as well as reservoirs of adaptive genetic variation critical for evolutionary response to changing climate (Kramer and Havens 2009; Vitt et al. 2009, 2010).

## 3.5
## Key Findings

- Significant changes in a plant's vital rates may result in demographic and genetic changes at the population level. Resiliency of NTFP populations varies according to life history traits, as well as the ecological, management, and social context of harvest.

- Long-lived perennial plants tend to be highly sensitive to decreases in adult survival from their harvest or the harvest of parts (e.g., bark, roots, rhizome, bulb), and may have significant negative impacts on long-term population persistence.

- Long-term harvest sustainability operates across multiple scales and levels of ecological organization, requiring understanding of processes that go beyond individual populations and species.

- Crafting plans for sustainable management requires understanding the effects of harvest over time and across landscapes, as well as the impacts of climatic variability and change.

- Species response to disturbance is determined by reproduction mechanisms and responses to changes in light or water availability, substrate condition (e.g., mineral versus organic soils), and nutrient availability, as well as other edaphic factors.

- Most ecological studies have focused on harvest sustainability at the population level and effects on biodiversity, community interactions, and ecosystem functions have received insufficient study.

- NTFPs include a diverse array of species that span broad taxonomic and environmental boundaries which makes assessing the vulnerability of them to climate change a challenge, especially demographic, evolutionary, and spatial responses to climate change.

- Climate-change integrated forest management requires a wide range of strategies that mitigate stressors, increase forest resilience, and conserve habitat refugia and corridors to allow for responses to range shifts and human-assisted relocation of species that are unable to shift ranges.

## 3.6
## Key Information Needs

- A comprehensive understanding of the ecological impacts of harvesting and stewardship.

- Improved maps and other geospatial tools of NTFP species ranges.

- Accurate estimations of growth, yield, and mortality that allow sustainable harvest levels and practices in natural and forest farmed settings, including many types of plant organs (e.g., roots, leaves, fruit).

- Monitoring more populations over longer periods and testing of different management practices.

- Improve abilities to evaluate the impacts on native NTFP species and populations in the context of potential drivers of population change (e.g., habitat conversion, invasive species, pollinators, fire, herbivory, drought) relative to climate change.

- Improve modeling of projected impacts on NTFP species and populations from climate change and variability that would support risk assessments for NTFP species leading to mitigation and adaptation strategies.

- An understanding of potential impacts at all ecological levels, from species to landscapes, that consider ecological interactions to sustain populations, with clear understanding on how NTFP species contribute to forest health and resiliency to climate and other factors.

- Information from traditional, local, and science-based knowledge sources that are integrated fully into comprehensive management strategies.

## 3.7
## Conclusions

Hundreds of NTFPs are harvested in the United States and its affiliated territories, but ecological studies exist for relatively few of these. Despite current overharvest of some species, reviews and meta-analyses suggest potential for sustainable harvest of many (Schmidt et al. 2011, Stanley et al. 2012, Ticktin 2004). The challenge lies in identifying and meeting the conditions necessary to long-term sustainability. Because data are lacking on a majority of NTFP species, research on basic ecologies, harvest dynamics, harvester stewardship practices, and production and market dynamics are needed.

Basic ecological information needs include reproductive biology, habitat requirements, and response to disturbances. Data on interactions between pollinators and seed dispersers will be needed to sustain NTFP populations and understand their contributions to the resilience of ecological communities. Improved mapping of NTFP species ranges, monitoring of populations over longer periods, and testing of management practices are needed to measure variation over time, space, and management strategies. Examining population recovery rates from seeds and resprouts in different habitats is a priority for species harvested for their underground organs.

Understanding harvest dynamics and harvester stewardship practices is critical and will require studies of how, why, and where harvesters gather, including traditional and local ecological knowledge (Baron et al. 2015). Analyses of the relative effects of harvest versus other potential drivers of population decline (e.g., habitat destruction, invasive species, fire, herbivory, drought), as well as potential synergistic effects, is a priority. Best practices for such research and its use as a basis for management plans includes long-term participatory collaboration with harvesters and managers (Cudill and Rodela 2012).

Production and market studies also are essential to understanding the long-term sustainability of commercially traded NTFPs. Research on the volume and timing of trade will benefit basic ecological and harvest dynamics research alike, as will production analyses examining the absolute and distributional costs and benefits of wild harvests versus cultivation. Phytochemistry research may support improved sustainability by identifying options for substitution of plant materials with lesser impacts on species and community viability and determining whether harvest practices affect the presence of active medicinal compounds.

The potential vulnerability or resilience of NTFP species can be identified based on ecological, social, and market characteristics of species, habitats, and harvest systems. For species potentially at risk, there are multiple approaches to assessing and identifying sustainable harvest, depending on the needs, time, and resources available (see Cunningham 2001, Schmidt et al. 2011, Ticktin 2015, for guidelines on what methods to use, and when). Demographic models provide powerful tools for evaluating the effects of harvest and other pressures on long-term population growth rates (Crone et al. 2013, Ellner and Rees 2006). Risk assessments for key species would include identification of life history traits linked to vulnerability; assessments of how harvest interacts with changing climate to affect viability of NTFP populations; impacts of harvest on local genotypes and associated implications for adapting to climate change; and, evaluation of methods for, and relative benefits and risks of, managed relocation, including cultivation.

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