



## REDUCED NATURAL FOODS ALTER BOTTOM-UP PRESSURES ON AMERICAN BLACK BEARS

Spencer J. Rettler<sup>1,2</sup>, David L. Garshelis, Andrew Tri, Véronique St-Louis, and James Forester<sup>1</sup>

### SUMMARY OF FINDINGS

For bear management in North America, food availability (bottom-up) and hunting pressure (top-down) are the primary factors that limit population growth. Whereas seasonal and year-to-year variation in production of fruits and nuts (i.e., primary bear foods) have been widely recognized, long-term trends in natural food production have rarely been reported. Here we compared the current (2015–2017) availability of 18 fruits and nuts constituting the main foods of American black bears (*Ursus americanus*) in north-central Minnesota to what was available during the 1980s. The study area was the same in both time frames, constituting primarily national forest, where timber harvesting was routine in the 1980s but much less so in recent decades. We hypothesized that forests matured over the 25-year period and consequently produced less fruit for bears. Within each of 12 forest types, we measured the abundance, productivity and biomass (kg/ha) of each fruit-bearing plant species using the same methodology as used in the 1980s. For all groups of species, the probability of forest stands producing any fruit at all in the recent time period was half (~40%) what it was in the 1980s (~80%). At the landscape scale, we estimated a ~70% decline in biomass availability due to a reduction in young forest types that produce the most fruit, and also a ubiquitous decline in fruit production across most forest types. Our hypothesis that this decline was due to reduced timber harvesting was only partially correct, as we observed the same decline in fruit production within mature forest of the same type and canopy closure, and also along edges of stands with high light penetration. Earlier springs, with potentially greater vulnerability of flowers to frost, and an ongoing invasion of invasive earthworms, which are known to radically affect the soil, may be additional causes for diminished fruit production. Bears will likely need to alter their foraging habits to compensate for reduced availability of natural foods as the landscape continues to change. This study demonstrates the complexity of forest management and its unintended effects on wildlife.

<sup>1</sup> Department of Fisheries, Wildlife and Conservation Biology, University of Minnesota, St. Paul.

<sup>2</sup> This project served as an M.S. thesis, completed in June 2018,

## INTRODUCTION

In forested environments, different combinations of overstory and understory species compositions, tree densities, and canopy closures will provide distributions of food and cover that favor different wildlife species. Here we focus on how these forest attributes can affect the production of food for an omnivorous, but largely frugivorous big game species, the American black bear (*Ursus americanus*; henceforth black bear). This forest-dependent species was historically over-exploited by hunters; however, it is now increasing numerically and expanding geographically across most of its range in the USA, Canada, and northern Mexico (Garshelis et al. 2016). The widespread increase in this species is mainly attributable to purposeful control of top-down forces, specifically reductions in legal and illegal hunting and killing of nuisance animals. Also, in some areas, forest cover has expanded, generally promoting increased vitality of this species.

Food availability affects bear body mass and condition, which greatly influences reproduction (Noyce and Garshelis 1994, Samson and Huot 1995, Costello et al. 2003); this can significantly affect the resilience of the population to hunting (Kontio et al. 1998). Black bears in Minnesota rely on over a dozen key fruits, which are produced on bushes or small trees in the forest understory from June through October (Garshelis and Noyce 2008). Habitat-specific abundance (areal coverage) and productivity (fruit production) of the key bear foods was measured on one study site near the center of the Minnesota bear range from the mid-1980s through the early 1990s. A notable finding was that fruit biomass for bears was especially high in young, regenerating or planted forests (Noyce and Coy 1990). Annual timber harvest on this study site declined from roughly 70,000–100,000 board-feet during the mid-1980s to mid-1990s to 20,000–60,000 during the early-2000s to 2015 (United States Forest Service 2017). Because of this reduced forest cutting, forest composition changed and food-rich stands of young aspen and young pine plantations, which were common in the 1980s, matured and appeared to shade out some of the fruit-producing understory. We hypothesized that fruit production declined, and that this could negatively affect the bear population.

## OBJECTIVES

1. Determine whether declines in bear food production occurred since the 1980s.
2. Quantify the magnitude of change in bear food production since the 1980s, and identify probable causes for change.

## STUDY AREA

Located in northcentral Minnesota, the Chippewa National Forest (CNF) falls in the transition zone between the boreal forests to the northeast and the temperate forests in the central part of the state. Our study area covered 62,200 ha, dominated by the eastern extent of the CNF (42% of study area); the remainder included some of the George Washington State Forest (11%), county land (6%), private land (18%), commercial timber industry land (8%) and open water (15%) (Figure 1). Within this area were both lowland and upland forests. Lowlands were dominated by one or more of these species: speckled alder (*Alnus incana*), black spruce (*Picea marina*), tamarack (*Larix laricina*), black ash (*Fraxinus nigra*), northern white-cedar (*Thuja occidentalis*), quaking aspen (*Populus tremuloides*) and balsam poplar (*Populus balsamifera*). Upland forests were dominated by some of these species: quaking and bigtooth aspen (*Populus grandidentata*), maple (*Acer spp.*), red pine (*Pinus resinosa*), paper birch (*Betula papyrifera*), and balsam fir (*Abies balsamea*). From 1990 to 2009, 2% of federal land, 4% of state land and 6% of county land was disturbed (e.g., timber harvest, natural blowdown from storms; Garner et al. 2016). From 2010 to 2015, timber harvest activity on state and county forests (9%) remained significantly higher than in the CNF (3%). Many lakes, forest roads, and recreational trails can

be found throughout the public land. This area was heavily hunted for bears throughout the scope of this study (1980s to present) due to the large extent of easily-accessible public land.

## METHODS

We estimated abundance and production of 18 species of wild fruits (or groups of related species; Table 1) during early July to late August, the peak fruiting season. The fruit sampling methodology we used during 2015–2017 (n=126 unique forest stands) was patterned after that used by Noyce and Coy (1990), who sampled the same study area during 1984–1989 (n=142 unique stands). We designated 12 forest stand types based on the species composition and age category of dominant trees: mature aspen (>30 years old), aspen regeneration (5–15 years), lowland aspen (mixed with balsam poplar), mature red pine (>35 years), young pine plantations (8–20 years), maple, paper birch, recent timber harvests/upland brush (<5 years), black ash, lowland brush, northern white-cedar, and black spruce/tamarack. Initially, we chose stands to sample based on forest stand inventory data derived from public land spatial databases. Before sampling, however, we confirmed the stand type on the ground. We attempted to sample 10 stands of each forest type, but in some cases we could not locate that many. Furthermore, we attempted to resample the same stands in different years, but had to find alternate ones if the target stand had been harvested since the first sample.

Within each sampled stand, we temporarily marked the boundaries of 12 circular plots, each with a radius of 3 m. Plots were spaced 30 m apart in a rectangular grid (3 x 4 plots). For small stands, we used closer spacing. For stands with an obvious edge (e.g., abutting a forest road or trail), we positioned one row of plots along the edge in order to discern whether fruiting was enhanced by the increased sunlight. During the 1980s, only interior plots were sampled with the exception of a few stands during the 1988 field season. We measured canopy cover (%) within each plot, including all vegetation >2 m tall and above the fruit-bearing species, using spherical convex densitometers or by visual inspection. We calibrated visual inspections among observers and with densitometers.

We ranked abundance of fruit-producing species on a 0–4 scale based on areal coverage within the plot: 0 = 0%, Trace = 0.1–0.9%, 1 = 1–5%, 2 = 6–33%, 3 = 34–67%, 4 = 68–100%. We used the mid-point percent value for further treatment of these data (e.g., abundance rating of 2 was converted to 19%). We separately ranked fruit production, also on a 0–4 scale: 0 = little to no fruit, 1 = below average production, 2 = average, 3 = above average, and 4 = bumper crop. We considered only the number of fruits produced (per plant or per area), not their size or ripeness. To ensure that our ratings matched those of Noyce and Coy (1990), we developed our subjective scale using their fruit yield data (range of values for fruits/m<sup>2</sup> corresponding to each subjective rating). We also used their estimate of mass of each fruit (when fully ripe) to convert our subjective ratings to biomass in grams per m<sup>2</sup>.

A description of the statistical procedures for analyzing data and results of statistical tests are provided in Rettler (2018). For some analyses we compared species that ripened in summer versus fall, or whether they were produced on a short versus tall shrub (Table 1).

We could not directly compare landscape level changes in fruit availability across decadal periods because forest type composition in the 1980s was not available, and forest type information for 2015–2017 was available only for public lands (federal, state, and county, comprising 69% of the land area of the study area). For the 1980s, we used visual estimates of forest composition within 500 m of telemetry flight locations of radiocollared bears, and for the 2010s we used the same set of locations, but instead of visually identifying forest types from an airplane, we used stand inventory layers for public lands. More details are provided by Rettler (2018).

## RESULTS

### Plant Abundance

Five fruiting species declined in abundance between the 1980s and 2010s. The most notable decline was for sarsaparilla in aspen regeneration (12% decline) and paper birch (11% decline). Abundance of beaked hazel and downy arrowwood increased between the sampling periods. The largest increase was for beaked hazel in mature aspen (13%). Notwithstanding these species-specific changes, the overall abundance of plant species groups (Table 1) that produced fruit or nuts (during the summer, fall and on short or tall shrubs) did not change between the 1980s and 2010s.

### Fruit Production

Ten species declined in fruit production between the 2 decadal sampling periods; no species showed an increase. All species groups had a higher probability of producing some fruit in the 1980s (~80%) than in the 2010s (~40%; Figure 2). However, when fruits were present, there was no difference in production ratings. Increased canopy cover reduced the probability of short shrubs and summer food species producing fruit.

Edges and interior parts of stands had similar probabilities of producing some (non-zero) fruit. However, when fruits were present, edges produced more fruit. The edges of stands in the 2010s had a lower probability of producing fruit than the interior of stands in the 1980s, but there was no difference in fruit production ratings on the edges of stands in the 2010s versus the edges in the 1980s.

### Fruit Biomass

Half (9 of 18) of the fruiting species sampled declined in biomass from the 1980s to 2010s (Figure 3). The difference in biomass of fruits between the 1980s and 2010s was due to the higher probability of stands producing no fruit during the 2010s, not the biomass of fruits that were present. As canopy cover increased, fruiting biomass declined. Canopy cover significantly increased from the 1980s to 2010s in black ash, mature aspen, lowland aspen–balsam poplar, and mature red pine, and declined in upland brush (Figure 4).

### Landscape Biomass Availability

The availability of mature aspen on public lands increased from 28% of the study area in the 1980s to 49% in the 2010s, whereas young aspen correspondingly declined from 18% to <1% over these 3 decades. Pine plantations showed the same trend—young plantations comprised 3.4% of the landscape in the 1980s versus <1% in the 2010s, whereas mature pine increased from 4% to 8%. The reduction of young aspen and young pine on the landscape yielded a loss of >900,000 kg of bear foods on the study area, and the reduction in birch caused an additional loss of >300,000 kg from 1980s to 2010s (Figure 5). Combining all forest types, the total loss of biomass was estimated at >1.5 million kg for public lands on the study area (~37,000 ha). We estimated that state and county lands produced a higher biomass of bear foods due to shorter timber cutting cycles producing more regenerating aspen, upland brush, and pine plantations.

## DISCUSSION

We observed a large (~70%) reduction in summer and fall fruit availability within forest types across the landscape from the 1980s to 2010s, which would certainly be sufficient to affect the biology and behavior of bears. The best fruit production years in the summer and fall of the 2010s were similar to food failures in the 1980s. This decline in food availability over 3 decades occurred across many forest types and was driven by 2 factors: (1) changes in composition of the landscape, with a reduction in young, productive forest types, and (2) reduced productivity in

stands of the same age and type. For the most part, it was not driven by a decline in abundance of fruit-producing plants. Most fruit-producing plants in northern Minnesota were large, woody shrubs or small trees, which would not change much in abundance without a significant disturbance. Conversely, short herb and shrub species whose vegetative structures die back and regrow each year showed a greater change in abundance. Most notable of these were sarsaparilla and raspberry.

Sarsaparilla is typically the first fruit-bearing species to ripen in the summer and was one of the most common berries found in bear scats during the 1980s (Garshelis and Noyce 2008). Since then, its abundance, production and fruit biomass have declined precipitously in many of the forest types in which it was found. Likewise, raspberry was the single largest contributor to overall fruit biomass in younger forests in the 1980s (Noyce and Coy 1990), but it has drastically declined over the past few decades. Thus, bears in this area have lost 2 early-summer staples in their diet. Additionally, several important late-summer and fall foods have declined, notably blackberry, chokecherry, and beaked hazel. During the 1980s, blackberry and chokecherry contributed the most late-summer fruit biomass in upland and lowland forests, respectively (Noyce and Coy 1990). Now these species barely contribute to the food availability in aspen regeneration, pine plantations and lowland aspen. The reduction in beaked hazel production is especially striking because this is the most important fall food across much of Minnesota, and so its abundance can strongly affect bears' vulnerability to hunting (which is done using bait) and their condition before entering dens (Garshelis and Noyce 2008, Noyce and Coy 1990).

Our results indicate that a prolonged reduction in timber harvest activity in the CNF from the mid-1990s to 2010 led to a change in forest structure (i.e., increased canopy cover), an increase in mature forest types on the landscape, and consequently a reduction in food availability. Most of the mature forest stands were >40 years old; therefore, they were in the understory reinitiation phase of stand development after timber harvest. Competition for sunlight is high during this time and understory tree species begin to fill in new gaps as the dominant tree species naturally thins. Maple was often present in the understory and mid-canopy of mature aspen, paper birch and red pine stands, and new cohorts of black ash filled in the mid-canopy in lowland aspen and black ash stands. This likely explained the increase in canopy cover of these mature forest types since the 1980s, which lowered the productivity of many of the shade-intolerant fruiting species. However, this relationship does not explain the large reduction in food availability in young forests, as well as why most of the stands in the 2010s did not produce any fruit or nuts (Figure 2). Furthermore, while the edge of forests did have higher productivity than the interior during the 2010s, due to greater light penetration, the probability of any fruit production on the edges of forests during the 2010s was still much lower than the interior of forests during the 1980s.

To investigate why many young forests and forest edges with plentiful light did not produce any food in the 2010s we considered the hypothesis that climate warming shifted the phenology of flowering to earlier in the spring and increased their exposure to damage from a later frost. While we know that killing frosts occasionally destroy berry crops in Minnesota, we did not find any evidence from the available climate and phenology data (NOAA 2018, NPN 2018) to indicate that this was more common in the 2010s than the 1980s. However, without detailed historical and contemporary information on plant phenology in relation to weather, we cannot conclusively determine whether the flowering dates have changed or whether the frequency of frost events affected fruit production.

Another hypothesis is that invasive European earthworms may be affecting fruit production. There is clear evidence that earthworms have increased in the CNF, and their infestation of the forest was still in process decades ago; because the worms were introduced from fishing bait, their colonization is strongly associated with the many fishing lakes, cabins, and road access in

the CNF (Holdsworth et al. 2007a). In fact, the CNF is the most road-dense national forest in the country, and our study site has a particularly high density of small- to medium-sized lakes accessible for fishing (Figure 1). It is known that earthworms have a major effect on the physical, chemical, microbial, and mycorrhizal composition of the soil, and this significantly reduced abundance and diversity of understory plant species in the CNF and other northern temperate forests (Bohlen et al. 2004, Hale et al. 2006, Holdsworth et al. 2007b). The large decline in abundance of wild sarsaparilla in the CNF may be explained by the rarity of this species in areas heavily infested by earthworms (Frelich et al. 2006, Holdsworth et al. 2007b). We have not seen studies relating abundance of invasive earthworms to fruit production, but this seems like a logical extension given the reduction in soil nutrients such as nitrogen and phosphorous, and consequent stress to rooting systems following an earthworm invasion (Dobson et al. 2017, Frelich et al. 2006). If this hypothesis is correct, it may mean that the decline in bear foods coincides with earthworm infestations, which are much more widespread in Minnesota forests than just the CNF. Also, as noted by Frelich et al. (2006), the relationships between earthworms and forest productivity may be exacerbated by other agents, such as deer browsing and climate change, and climate warming is likely to exacerbate the earthworm invasion in closed canopy forests (Eisenhauer et al. 2014).

This reduction in food resources could have profound impacts on bears' body condition, reproduction, cub mortality (starvation), adult mortality (hunting vulnerability), and ultimately the carrying capacity of the landscape. In an analogous case on an island in Washington state, heavy logging promoted a rich pulse of fruiting, which supported a vigorous black bear population; when logging ceased, the bear population crashed (Lindzey et al. 1986).

When natural foods are scarce, bears often seek calorically-dense anthropogenic foods such as garbage and crops, bringing them into proximity to humans and increasing their chance of being killed. For bear populations that are harvested, there is strong evidence that years of poor food availability results in more visitations to hunters' baits and, consequently, increased hunter success (Noyce and Garshelis 1997; Obbard et al. 2014). In Minnesota, hunting success has increased dramatically in recent years, which may in part be due to bears' increased reliance on hunters' baits (Garshelis and Tri 2018). Thus, there is an interaction between top-down and bottom-up pressures, which may amplify the negative impacts on the population.

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Table 1. Fruits commonly consumed by bears in north-central Minnesota and sampled in this study (1980s and 2010s), grouped by season of ripening and structure of plant.

Common name	Scientific name	Summer food	Fall food	Short shrub	Tall shrub
Wild sarsaparilla	<i>Aralia nudicaulis</i>	X		X	
Currant	<i>Ribes spp.</i>	X		X	
Gooseberry	<i>Ribes spp.</i>	X		X	
Blueberry	<i>Vaccinium spp.</i>	X		X	
Red raspberry	<i>Rubus idaeus</i>	X		X	
Alder-leaved buckthorn	<i>Rhamnus alnifolia</i>	X		X	
Juneberry	<i>Amelanchier spp.</i>	X			X
Pin cherry	<i>Prunus pensylvanica</i>	X			X
Choke-cherry	<i>Prunus virginiana</i>	X			X
Blackberry	<i>Rubus allegheniensis</i>		X	X	
High-bush cranberry	<i>Viburnum trilobum</i>		X		X
Downy arrow-wood	<i>Viburnum rafinesquianum</i>		X		X
American hazel	<i>Corylus americana</i>		X		X
Beaked hazel	<i>Corylus cornuta</i>		X		X
Pagoda dogwood	<i>Cornus alternifolia</i>		X		X
Red-osier dogwood	<i>Cornus sericea</i>		X		X
Round-leaved dogwood	<i>Cornus rugosa</i>		X		X
Wild plum	<i>Prunus americana</i>		X		X



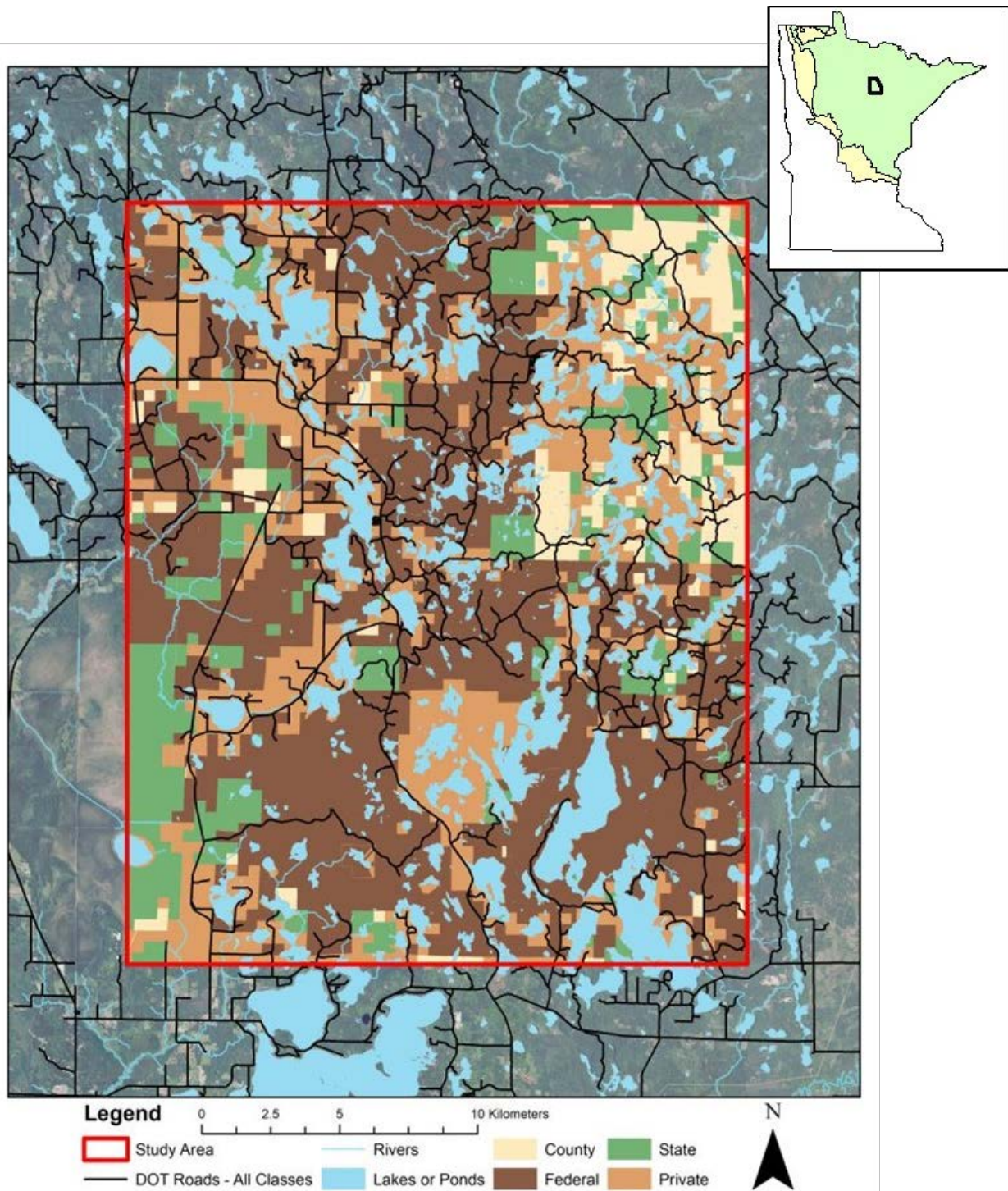


Figure 1. The study area, located in the center of primary bear range in Minnesota (green area in upper right inset) is largely public land (dominated by the Chippewa National Forest), with a high density of roads and lakes.

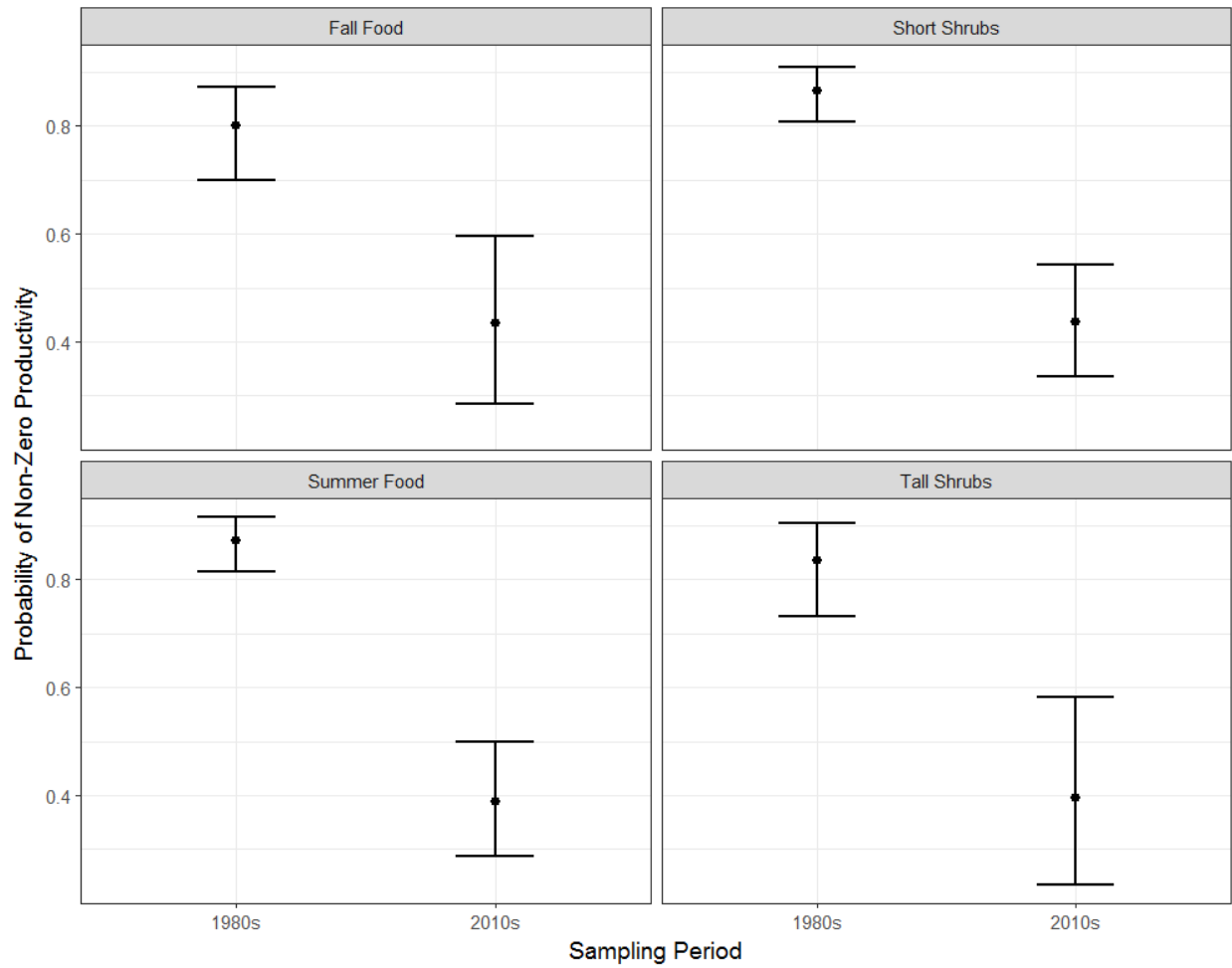


Figure 2. Probability of some fruit production (i.e., non-zero production) declined from  $\geq 80\%$  in forest stands in the 1980s to  $\sim 40\%$  in the 2010s for summer and fall bear foods, and short and tall shrubs (or small trees) producing bear foods (i.e., results were the same seasonally and for all plant types) in the study area in Minnesota.

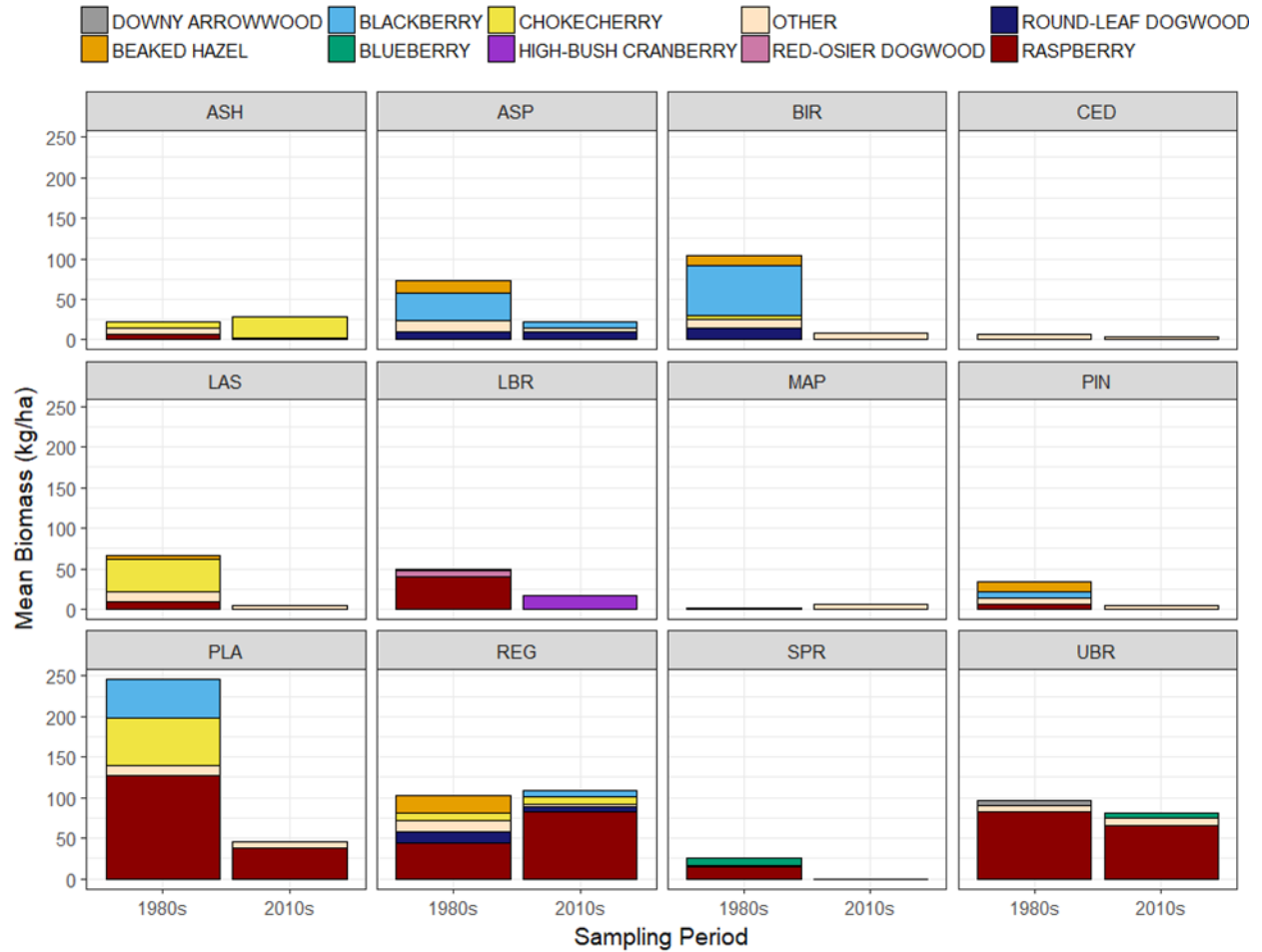


Figure 3. Mean biomass of bear foods produced in different types of forest stands in the CNF study area in Minnesota during the 1980s versus 2010s. ASH=black ash, ASP=mature aspen, BIR=birch, CED=white cedar, LAS=lowland aspen, LBR=lowland brush, MAP=maple, PIN=red pine, PLA=pine plantation (8–20 years old), REG=aspen regeneration (5–15 years old), SPR=spruce-tamarack, UBR=upland brush/clearcuts.

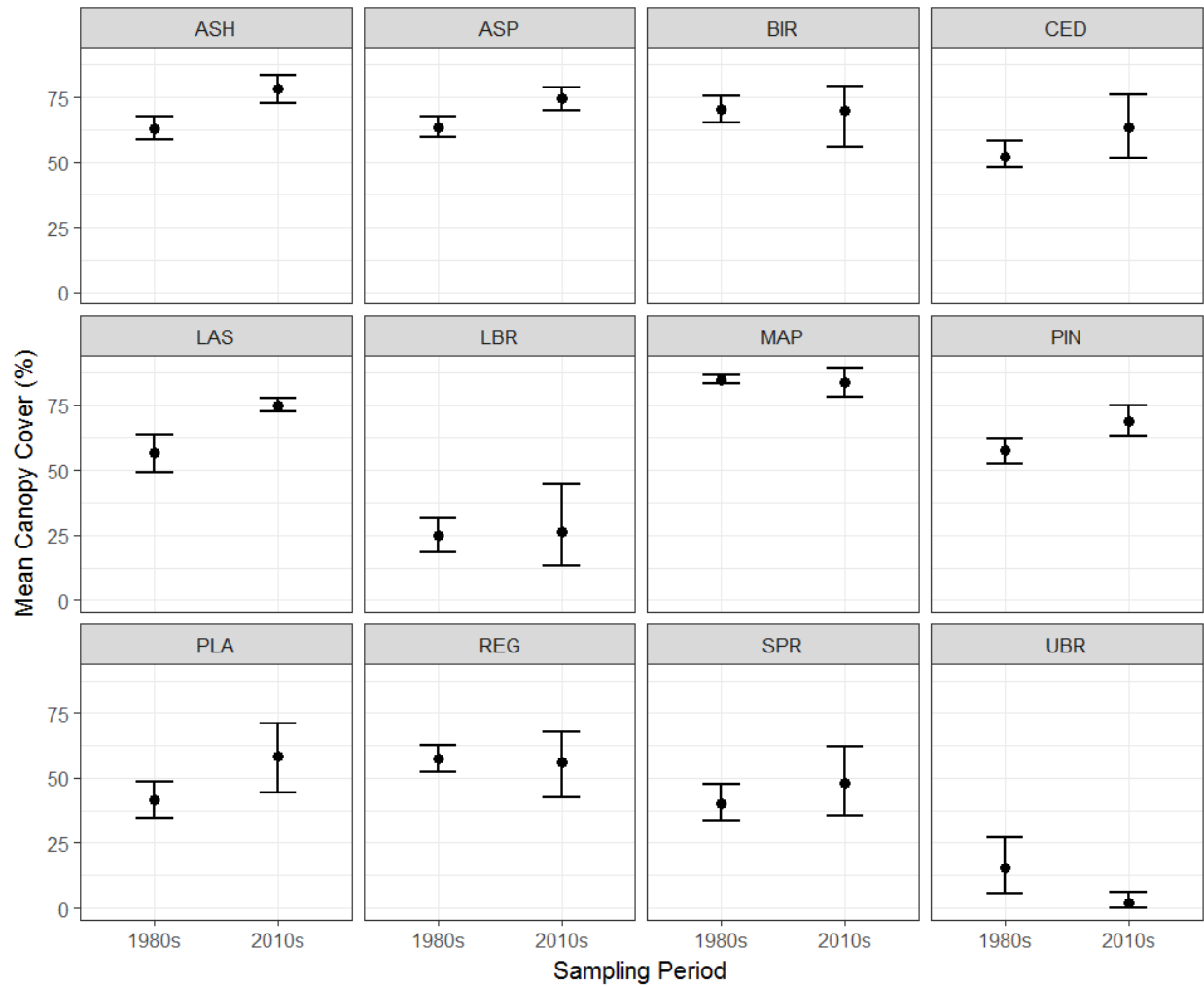


Figure 4. Mean canopy cover in different forest types (abbreviations as in Figure 3) in the CNF study area in Minnesota during the 1980s versus 2010s. Error bars depict 95% bootstrapped confidence intervals.

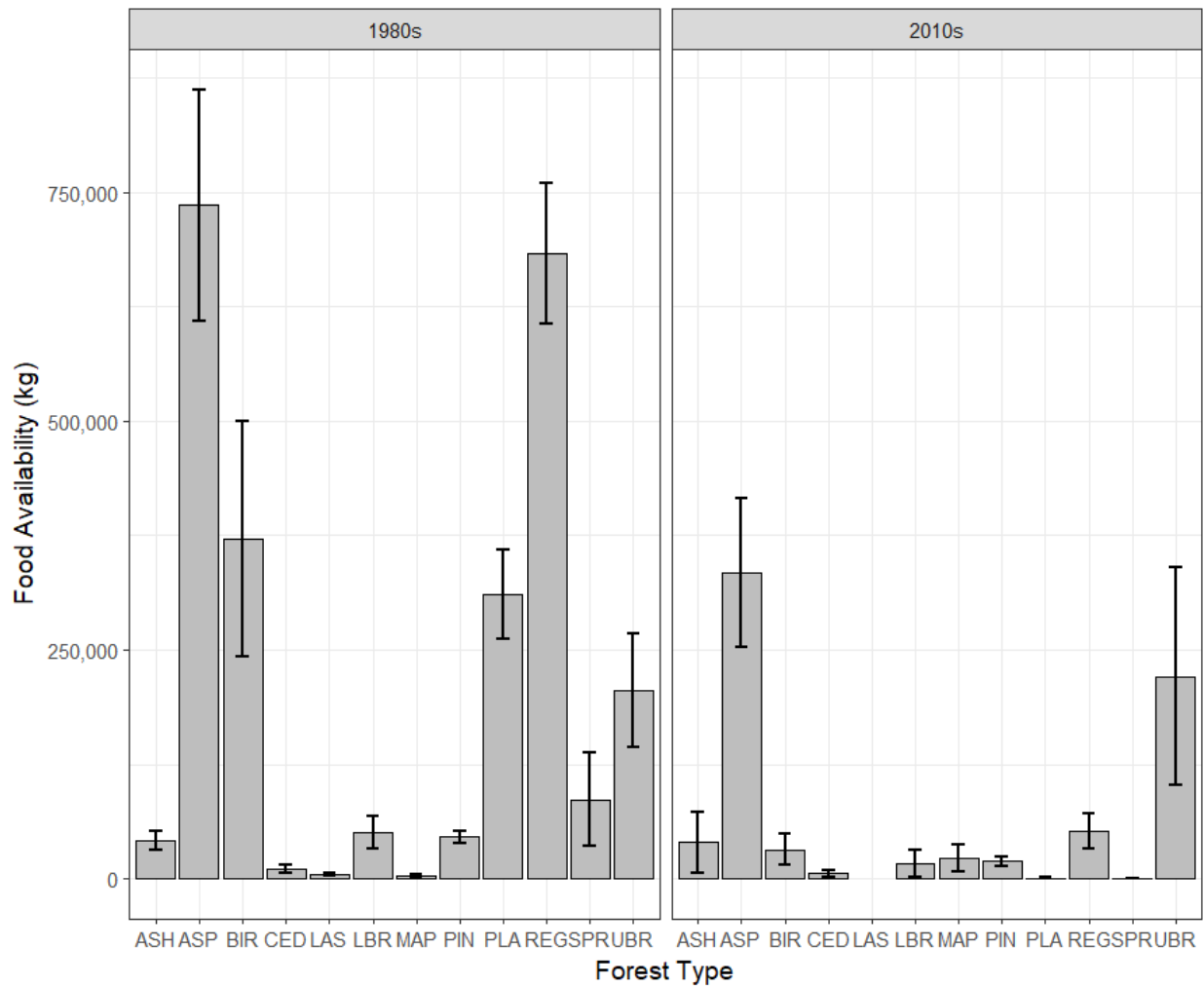


Figure 5. Estimated biomass of fruits important in the diet of bears in the CNF study site in Minnesota (estimated for public lands: 37,000 ha) during the 1980s versus 2010s in different types of forest stands (abbreviations as in Figure 3). The change from the 1980s to 2010s reflect a combination of (a) changes in amount of each type of forest in the study area, (b) changes in amount of fruit-producing plant species, and (c) changes in fruit production.