

Appendix A

Description of model parameters and equations, including differences from the Person (2001) model, where applicable.

Deer model

The deer model we constructed was not sex or age-structured, and included only an adult segment of the population with characteristic survival and reproductive rates (i.e., all deer were considered to be equally vulnerable to mortality causes, and sex ratio was effectively 1:1). Person (2001) also employed this model structure, after exploring sex- and age-structured models and finding no difference in model performance or outcomes (Person 2001, p. 54). We assumed density-dependence recruitment, which we approximate using a theta-logistic function. Density dependence in deer has not been specifically quantified in this ecosystem, and as a result, we retain this relationship from Person (2001). However, we add predation by black bears of fawns and adults based on recent work on deer ecology conducted on Prince of Wales Island (Person et al. 2009; Gilbert 2015). The parameters of the deer model describe U_t , the deer population in spring (prior to the annual birth pulse). The starting number of deer in year 1 in each wolf pack area was set as a proportion relative to nutritional carrying capacity (K_i ; see below), with the proportion generated from a uniform distribution ranging from 0.5–1. The deer population at time t is calculated across pack areas as $U_t = \sum_{i=1}^j U_{t(i)}$, where j is the number of wolf packs. For each pack area i , the deer population at time t is calculated as:

$$U_{t+1(i)} = U_{t(i)} + R_{ut(i)} - BA_{t(i)} - CP_{at(i)} - H_{t(i)} \quad (\text{A.1})$$

where $R_{ut(i)}$ is recruitment into the deer population, $BA_{t(i)}$ is predation mortality of adult deer by black bears, $CP_{at(i)}$ is predation mortality of deer by wolves, and $H_{t(i)}$ is death from human hunting. C , The per-capita wolf predation rate (26 deer/year/wolf, equivalent to 77% deer in the diet), was based on scat dietary analysis in the original model (Person et al. 1996), p. 42), and we updated this rate to 15 deer/year/wolf based on stable isotope analysis indicating that deer comprise approximately 44% of the diet of wolves on Prince of Wales Island (Szepanski et al. 1999). Wolf diets were randomly generated per pack per year from a normal distribution, with a mean of 15 and an SD of 4 (Person 2001). Further, we truncated wolf diets so that they ranged from 10-100% deer (i.e., 3.4–34 deer/wolf/year), to ensure that wolves used at least some deer each year.

Recruitment into the deer population is described as:

$$R_{ut(i)} = [1 - BF_{t(i)}] U_{t(i)} r_{max} \left[1 - \left(\frac{U_{t(i)}}{K_{t(i)}} \right)^\theta \right] \quad (\text{A.2})$$

where $R_{ut(i)}$ is recruitment in pack area i at time t , r_{max} is the maximum per capita rate of increase in the absence of predation and hunting, θ is the density dependence parameter, $K_{t(i)}$ is the carrying capacity of deer in pack area i at time t , and $BF_{t(i)}$ is predation of fawns by black bears. In addition, severe winters with deep snow are known to strongly impact deer populations in Southeast Alaska, primarily through reduced fawn survival (Person et al. 2009; Gilbert 2015). Consequently, we included a binomial parameter for

winter severity, the derivation of which is described in more detail in subsequent sections. While adult deer can die during extremely severe winters, we chose to include the effects of severe winters on deer by reducing recruitment to zero if a severe winter occurred. This is a relatively optimistic assumption from the standpoint of deer abundance, as while recruitment is unlikely to be truly zero (Gilbert 2015), adult deer also die during severe winters in reality (Klein & Olson 1960; Kirchhoff 1994; Person et al. 2009), while we do not include any adult deer mortality due to severe winters. We assumed that bear predation of fawns was partially compensatory, so that as population density relative to nutritional K increased, proportion of bear predation that was compensatory increased, up to a maximum of 50% compensation. We used the following relationships to modify the base mortality rate of 0.46 ($SD=0.023$), from Gilbert (2015), by deer density:

$$\text{If } \left[1 - \left(\frac{U_{t(i)}}{K_{t(i)}} \right) \right] < 0.5, \text{ then } BF_{t(i)} = 0.5 * 0.46 \quad (\text{A.3})$$

$$\text{If } \left[1 - \left(\frac{U_{t(i)}}{K_{t(i)}} \right) \right] \geq 0.5, \quad \text{then } BF_{t(i)} = \left[1 - \left(\frac{U_{t(i)}}{K_{t(i)}} \right) \right] * 0.46$$

In contrast, we assumed predation of black bears on adult deer, as well as predation by wolves and death due to human hunting, were completely additive (Gasaway et al. 1992; Hayes et al. 2003), and thus constant despite changes in deer density. While it has been suggested that wolf predation on ungulates may follow a Type II functional response curve (Dale et al. 1994), no published relationships exists for a functional response curve between deer density and wolf predation. In addition, such a

functional response would likely have a large effect on deer only at very low deer densities (Dale et al. 1994). As a result, we treat wolf predation as a constant, density-independent rate, allowing us to simplify the model. In addition, we calculated predation by black bears on adults, $BA_{t(i)}$, as a rate of 0.03 ($SD = 0.0015$) based on analysis of combined deer mortality data from Person et al. (2009) and Gilbert (2015), following methods described in Gilbert (2015).

Predation mortality of deer by wolves, $CP_{at(i)}$, was the product of the average number of deer killed per wolf per year (C), and $P_{at(i)}$, the average number of wolves in pack i during year t ($P_{at(i)}$). $P_{at(i)}$ was calculated as the average (a) of the spring and fall wolf populations, $(P_{t(i)} + (P_{t(i)} + R_{t(i)})) / 2$, where $P_{t(i)}$ is the spring wolf population in pack area i and year t , and $R_{t(i)}$ is reproduction in pack i in year t .

Deaths of deer due to human hunting, $H_{t(i)}$, was represented as:

$$H_{t(i)} = (U_{t(i)} + R_{t(i)} + CP_{at(i)}) * h * (1 + \beta_d * Km\ Roads) \quad (A.4)$$

where h is the base rate of harvest when roads are absent and β_d is a coefficient representing the additional harvest of deer by hunters with an increase in road access. We assumed that wolf predation of deer occurred before hunting of deer by humans, although in reality some winter predation of wolves on deer would occur after the conclusion of the deer-hunting season. The values of h and β_d were taken from the published regression relationship in Person (2001, Page 80), based on harvest and road-length data from Prince of Wales Island. Specifically, (Person & Bowyer 1997) derived a baseline hunting rate, h ,

of 0.012, and regression relationships for additional risk in the presence of roads that was dependent on road length:

$$\text{Hunting rate}_{t(i)} = h * (1 + 0.038 * \text{Km roads}) \quad (\text{A.5})$$

Person and Bowyer's 1997 regression approach (pg. 23) to predict the hunting rate from baseline hazard modified by road density was to first regress total deer harvest in each wildlife analysis area (WAA) sub-area in GMU2 against road length, then convert to a risk model based on population of deer estimated from the USFS deer habitat suitability in each WAA. They used reported harvest data from GMU 2 from 1990-1995, which ranged annually from 2,466-3,227 deer (mean = 2,919). For comparison, between 1990-2020, the mean deer harvest in GMU 2 ranged from 1,746-4,257 (mean = 2,806, SD = 677). This approach does not account for road closure status, sightability differences by habitat type, or sex- and age- specific differences in habitat selection, hunting pressure and subsequent risk.

Deer nutritional carrying capacity, $K_{t(i)}$, depends on habitat in pack area i at time t , and is a product both of the underlying productivity of the area (i.e., what original old-growth forest types existed in the area), and of subsequent timing and extent of timber harvest. Following timber harvest, forage changes in predictable ways through stages of forest succession (Alaback 1982), and as a result carrying capacity of deer changes as well (Hanley & Rogers 1989). The relationship between old-growth and second-growth forest types and deer carrying capacity is described in the model using the Deer Habitat Suitability Index (HSI), a system developed by management agencies in Southeast

Alaska to evaluate the effects of management decisions on deer (Suring et al. 1993). We chose to use the deer HSI as a metric of deer carrying capacity because it is the management standard in the region and is integrated into a GIS framework, and has yet to be replaced by a better solution.

Winter severity

In the model, deer recruitment, as mentioned above, depends on winter severity. Person (2001) treated winter severity as a random binomial variable based on temperature and precipitation data from the National Weather Service on Annette Island and Sitka, and data from Alaska Department of Fish and Game (unpublished data, Person 2001). This resulted in an average of 6 severe winters per century.

We re-defined a severe winter (i.e., a winter resulting in deer recruitment of zero) using precipitation data from Annette Island. We defined a severe winter as one in which total snowfall is >160 cm and maximum monthly snow depth is >25 cm. Parker et al. (1999) found that when maximum snow depths were >29.6 cm energy costs associated with movement of an average-sized deer (25–30 cm carpus height) increased significantly. During the first winter, fawns weigh ~40% less than adults (Parker et al. 1999) and their carpus height is ~10% shorter than for adults (Parker et al. 1984); thus, maximum snow depth of 25 cm should account for fawn energy expenditure with locomotion. For context, most forbs are covered when snow depths >10 cm (Parker et al. 1999). Applying the proposed definition, 2 winters of the 20 between 1995 and 2014 qualify as being “severe” (2000–2002, 2008–2009; Table 1). This rate of 10% is higher

than that used by Person 2001; 2 severe winters from 1947 to 1996), but is somewhat comparable with his estimate of 6 severe winters in a century (6%).

To predict future frequency of severe winters, we applied a predicted % change in snowpack provided by (Littell 2015) to our baseline probability of a severe winter (0.10). Littell (2015) predicted future snow depth between 2030 and 2059 using 5 different global climate models (GCM; Table A1); percent change in future snow depth ranged between 0 and -28.6% with a 5-GCM average of -19.2%. Applying this range of 3 percent changes to the baseline probability of a severe winter resulted in a predicted low probability of a severe winter (-28.6% change in snow fall) = 0.07 (0.10 rate X 0.714), a predicted average frequency (19.2% change) = 0.08 (0.10 rate X 0.808), and a predicted high frequency (0% change) = 0.10. These rates were applied as annual probabilities of a severe winter, depending on the scenario conditions.

Wolf Model

The wolf model is similar in structure to the deer model, with a non-age or -sex structured population of adult wolves and a density dependent annual reproductive output. In general, the wolf population takes the form:

$$P_{t+1(i)} = P_{t(i)} + R_{pt(i)} - T_{t(i)} - D_{t(i)} - M_{t(i)} + I_{t(i)} \quad (\text{A.6})$$

where $P_{t+1(i)}$ is spring pack size prior to parturition for wolf pack in area i . Starting wolf population size was randomly generated from the 2014 fall estimated wolf density and

standard deviation for Prince of Wales (Alaska Department of Fish and Game 2015), adjusted to match our study area, then reduced by the number of wolves reported harvested over winter 2014/2015 ($n = 7$), multiplied by a scalar for unreported harvest (31/18) based on Person and Russell (2008) and D. Person (pers. comm.). Next, a series of potential pack sizes was randomly generated from a mean value of 6 ($SD = 1$; (Person 2001), then allotted to wolf pack areas until the starting population size we had randomly generated previously had been reached, after which the remaining wolf pack areas received a starting pack size of zero. This ensured that while not all pack areas were occupied, both the size of individual packs and of the total population was realistic. In addition, pack sizes in all years was capped at 18 (as recommended by Alaska Department of Fish & Game personnel; Gretchen Roffler, personal communication). Pack members above this size threshold dispersed (see below). Further, the entire wolf pack dispersed if there were fewer than 50 deer in the pack area, to ensure that wolves in a specific pack area did drive the deer population in that area extinct in the model (given that deer can move between wolf pack territories freely in reality).

$R_{pt(i)}$ is recruitment to pack i and equaled:

$$R_{pt(i)} = b \left[1 - \frac{CP_{t(i)}}{\alpha U_{t-2(i)}} \right] \quad (A.7)$$

in the original model, where b is average litter size when ratio of prey:wolves is very high (i.e., $\frac{CP_{t(i)}}{\alpha U_{t-2(i)}}$ approaches 0), and α is proportion of deer population available to the pack in area i . Note that deer abundance is indexed to year $t-2$, which creates a 2-year lagged

response of wolves to deer. Proportion of deer available annually was a random variable drawn from a uniform distribution between 0.5-1.0. Average litter size was updated from 6 (SD = 0.3) to 4.1 (SD = 1.7) based on Person et al. (2009), and was randomly generated from a poisson distribution (Caswell 2001). Litter size was capped at 11 based on the recommendations from Alaska Department of Fish and Game Region I (Gretchen Roffler, personal communication).

$T_{t(i)}$ is the number for wolves harvested from pack i . Wolf harvest (harvest rate, HR , wolves harvest/100 km²) for 2015-2045 was a function of road density and distance from shoreline to nearest community since access to wolves was dependent on vehicles and boats (Person and Russell 2008, Table 5) :

$$HR_{t(i)} = [1.010 - 0.005(ocean\ distance) + 0.207(road\ density)]^2 \quad (A.8)$$

HR was converted to number of wolves harvested in each pack based on the size of the pack area and then multiplied by the unreported harvest scalar. These relationships were established by Person and Russell (2008), and were based on data on legal reported harvest of wolves on Prince of Wales from 1990-1998. As Person and Russell discuss, newer wolf harvest data were not included, because patterns of reporting of legal harvest changed after an emergency closure of harvest in 2001, while patterns of mortality in radio-collared wolves did not.

Number of dispersers from a pack in area i was a function of pack size and the predator:prey ratio:

$$D_{t(i)} = (P_{t(i)} + R_{pt(i)} - T_{t(i)}) \left[d \left(\frac{CP_{t(i)}}{\alpha U_{t-2(i)}} \right) \right] \quad (\text{A.9})$$

with a base dispersal rate (d) of 0.5 (SD = 0.3). We updated disperser annual survival probability to 0.34 (SD = 0.3) based on Person and Russell (2008) and we allowed the disperser pool to carry forward each year, which was not the case in the original model. The disperser pool from the previous year was harvested in the current year, via both legal and unreported harvest, based on harvest rates derived from Person and Russell (2008). Person and Russell (2008) found that survival differed between resident and dispersing wolves, but that cause-specific mortality proportions were not significantly different between these two classes. As a result, we first calculated the number of wolves in the disperser pool that died each year as $1 - \text{disperser survival}$. We then calculated number of dispersing wolves that died due to legal and illegal trapping based on ratios of cause-specific mortality rates reported for the entire population, as the disperser sample size was limited and therefore cause-specific mortality rates were not reported for the disperser class separately (Dave Person, personal communication). Person and Russell (2008) reported population-wide mortality rates of 0.23, 0.19, and 0.04 for legal harvest, illegal harvest, and chronic mortality respectively. We therefore calculated number of dispersers that died from legal harvest by multiplying by $(0.23)/(0.23 + 0.19 + 0.04) = 0.50$ (proportion of mortality due to legal harvest), and number that died from both legal and illegal harvest by multiplying by $(0.19 + 0.23)/(0.23 + 0.19 + 0.04) = 0.91$ (proportion of mortality due to harvest of all kinds).

A “cap” on trapping was implemented in some of the scenarios as a management action. To do this, we calculated the proportion of the previous fall’s total wolf

population (both residents and dispersers) trapped legally during the current year, which was realistic given that there will always be a 1-year time lag between the population census and subsequent management action. Then, if this proportion trapped legally exceeded the trapping cap proportion, the number of wolves predicted to be harvested that exceeded the trap cap were returned to the population in proportion to the relative contributions of the trapped residents versus dispersers. Then, returned resident wolves were re-distributed to each pack based on the weights calculated from the predicted harvest (i.e., packs with more predicted harvest experienced a reduced). These updates resulted in fewer unoccupied pack areas and pack areas unoccupied for a shorter period of time.

Number of wolves dying in pack area i in year t was a function of number of wolves in the area and the predator:prey ratio:

$$M_{t(i)} = (P_{t(i)} + R_{t(i)} - T_{t(i)} - D_{t(i)}) \left[m \left(\frac{CP_{t(i)}}{\alpha U_{t-2(i)}} \right) \right] \quad (\text{A.10})$$

with a base mortality rate (m) of 0.5 (SD = 0.3).

Immigration into a pack area ($I_{t(i)}$, 1 or 2 wolves) occurred if the pack area was occupied by <2 wolves, there were dispersers available to disperser from any pack area, and the predator:prey ratio ($\frac{CP_{t(i)}}{\alpha U_t}$) > 1/3 which represented sufficient prey to support 3 wolves. Unoccupied pack areas received 2 wolves when these conditions were met and those with 1 wolf received a single immigrant.

Table A.1. Historical, projected future, and projected percent change in precipitation as snow for Prince of Wales Island, Alaska based on five Global Climate Models (GCM; data from Littell 2015).

Climate conditions	Precipitation as snow (mm)	Change (%)
Historical (1970–1999)	453.3	--
GCM Projections (2030–2059 A2 emissions)		
UKMOHadCM3	453.3	0
CCCMA-CGCM3.1 t47	353.2	-22.1
MIROC3.2 medres	344.1	-24.1
GFDL-CM2.1	356.9	-21.3
MPI-ECHAM5	323.5	-28.6
5-GCM average	366.2	-19.2