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From fear to food: predation risk shapes deer behaviour, their resources and forest vegetation

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24 **Abstract**

25 The “ecology of fear” emphasizes the potential role of predation risk in shaping large herbivore
26 behaviour and the way they affect forest ecology. In this study we show how the presence or
27 absence of predation risk by hunters, together with or in the absence of carnivores, affect the
28 behaviour and ecological effects of Sitka black-tailed deer introduced to the islands of Haida
29 Gwaii, (British Columbia, Canada) or native to coastal BC. Deer in risk-free population showed
30 remarkable tolerance to human presence while deer exposed to severe culling in the recent past,
31 exhibited more costly anti-predator behaviors (long flight initiation distances and long travel
32 distances when fleeing; reluctance to consume foreign bait or to investigate baited traps; increased
33 night-time foraging) and were **more likely** to use exposed habitats. Contrasts in hunting histories
34 translated into dramatic variation in the nature, distribution and abundance of the understory
35 vegetation deer depended on.

36 The experimental translocation of unwary deer from an island without hunting to an island where
37 **culls had partially restored the vegetation, showed that the lack of costly anti-predator behaviors**
38 **was not significantly affected by the presence of abundant and higher quality forage.** We
39 interpreted these results as evidence that the experience of risk was key in explaining the observed
40 behavioral contrasts between deer populations with different risk histories. We strengthened this
41 conclusion by analysing the proportion of stable isotopes in deer bone collagen to show that deer
42 foraged less in the exposed intertidal zone when predation risk was higher.

43 Our results provide novel insights into how predation risk affects ecological networks, ecosystem
44 complexity and animal behaviour. By revealing the role of key species, they may enable better
45 strategies for future ecosystem restoration.

46
47 **Keywords:** ecology of fear, anti-predator behaviour, herbivore behaviour and predation risk, fear
48 and habitat selection, fear and behavioural change, fear and herbivore habitat effects

49

51 From its outset, Ecology was defined as the science of interactions (Haeckel 1866, Elton 1927).
52 Interactions were first assessed by observing direct relationships among species. Over time, the
53 increasing sophistication of empirical and experimental approaches improved our understanding
54 of ecological systems and revealed the importance and intricacies of indirect interactions. This
55 brought key insights into the central role that herbivores and their predators play in the structure,
56 function and stability of ecological systems [(Paine 1966, 1969, Estes et al. 2011) and review in
57 (Martin et al. 2020)]. Consequently, while the loss of species or populations across the world's
58 ecosystems is dire, the loss of species interactions is perhaps even more insidious, because often
59 unnoticed (Janzen 1974, Soulé et al. 2003, Valiente-Banuet et al. 2015). The loss (or addition) of
60 key players in the ecological fabric will affect, directly or indirectly, day to day processes in an
61 ecosystem, and, over time, potentially select for traits and behaviors better adapted to the new state
62 of the ecological network (Bøhn and Amundsen 2004, Ellers et al. 2012, Estes et al. 2013).

63 The key role animal behavior plays in species interactions and in ecosystems has been dramatically
64 highlighted by the emergence of the conceptual framework proposed in what was named the
65 “ecology of fear” (Brown et al. 1999). This framework focused on the profound non-consumptive
66 effect predator presence can have on prey behavior and on how prey interact with their habitat,
67 giving rise to the idea of a “landscape of fear”. It fostered predictions about the consequences risk
68 variation in space and time could have, not only on predator-prey interactions, but also on the
69 functioning of ecosystems at large (Palmer et al. 2022).

70 Focusing on ungulates, and deer in particular, Altendorf et al. (2001), building on the pioneering
71 work of Brown et al. (1999), predicted that, in presence of risk, animals should spend less time
72 foraging overall, and spent less time foraging at any given location. Presence or absence of risk
73 should therefore influence the avoidance or use of certain portions of their habitat, perceived either
74 as more exposed to, or as refuges from, predation (Williams et al. 2008, Bonnot et al. 2017).

75 In large areas of North America and Europe dramatic increases in large herbivore populations
76 resulted in the partial recovery of their main predators, wolves. In this context, concepts related to
77 an “ecology of risk” have created a need to better understand how predator-prey interactions affect
78 ecological systems and their conservation (Chitwood et al. 2022).

79 Furthermore, the increasing understanding of the complex range of interactions ungulates and
80 wolves have with human activities, such as farming, livestock husbandry, or forestry, opened new
81 research avenues to better understand the role of predation risk in mitigating the undesirable
82 ecological and societal impacts of large herbivores (Kuijper et al. 2016, Martin et al. 2020). Such
83 studies emphasized the need to better integrate the study of animal and ecosystem ecology with
84 the study of animal temperament and its evolution (Réale et al. 2007, Sih et al. 2012) if we want
85 to embrace the full complexity of natural systems (Chitwood et al. 2022).

86 In this study we tested how the effects of high deer abundance on the ecology of forests are
87 modulated by the presence ~~or absence~~ of hunting or predation. We took advantage of the
88 introduction, at the end of the 19th century, of Sitka black-tailed deer (*Odocoileus hemionus*
89 *sitkensis*) to Haida Gwaii, a remote archipelago in British Columbia, western Canada (Fig. 1),
90 (Golumbia et al. 2008). The archipelago provided islands with and without deer, and, where deer
91 were present, islands with and without a history of hunting by people. This arrangement provided
92 a quasi-experimental context in which to study how predation risk shapes deer behaviour, their
93 resources and forest vegetation by comparing islands with deer that varied in hunting history.
94 Studies on Haida Gwaii were compared with a mainland coastal site where deer have always faced
95 natural predators in addition to hunters (Darimont et al. 2007).

96 Specifically, we asked: (1) Does deer habitat use, shyness, avoidance, and activity rhythms vary
97 with the presence or absence of hunting or predation risk? (2) Do risk-related contrasts in deer
98 behavior modify how deer affect the forest ecosystem?

99 We show that risk histories were key in explaining the observed behavioral contrasts between deer
100 populations and their effects on forest ecology. These results emphasize the importance of
101 predation risk, and its loss, in the dynamics of ecological networks.

102

103

Material and Methods

104 Haida Gwaii and the study sites

105 Haida Gwaii is characterized by a humid temperate-oceanic climate, with mean annual
106 temperature of 8.5°C (Banner et al. 2014). Most of the archipelago is covered by temperate
107 rainforests dominated by western hemlock (*Tsuga heterophylla*), western redcedar (*Thuja plicata*),
108 and Sitka spruce (*Picea sitchensis*).

109 Native to coastal British Columbia, Sitka black-tailed deer were introduced to Haida Gwaii in the
110 late 19th century (Golumbia et al. 2008). They colonized all but a few small islands. The
111 occurrence of a few reference islands that never supported deer made it possible to demonstrate
112 that, on islands with long-term deer presence, independent of island size, deer herbivory was the
113 main factor structuring plant and animal communities (Martin and Baltzinger 2002, Gaston et al.
114 2006, Martin et al. 2010, Chollet et al. 2013), with important consequences on belowground
115 processes (Chollet et al. 2021a, Maillard et al. 2021). Recurrent experimental culls on two islands
116 allowed us to monitor the response of the aboveground vegetation and avifauna for 13 years
117 (Chollet et al. 2016).

118 The three islands in Laskeek Bay (52°53'12"N, 131°35'20"W) where we focused our study [Reef
119 249 ha, Kunga 395 ha, and East Limestone Island 48 ha] (Table 1) have all had deer present for
120 over 60 years at the time of study (Vila et al. 2004) and had no history of hunting (Fig. 1). Their
121 initial deer density was estimated at approximately 30 deer / km² (Daufresne and Martin 1997).

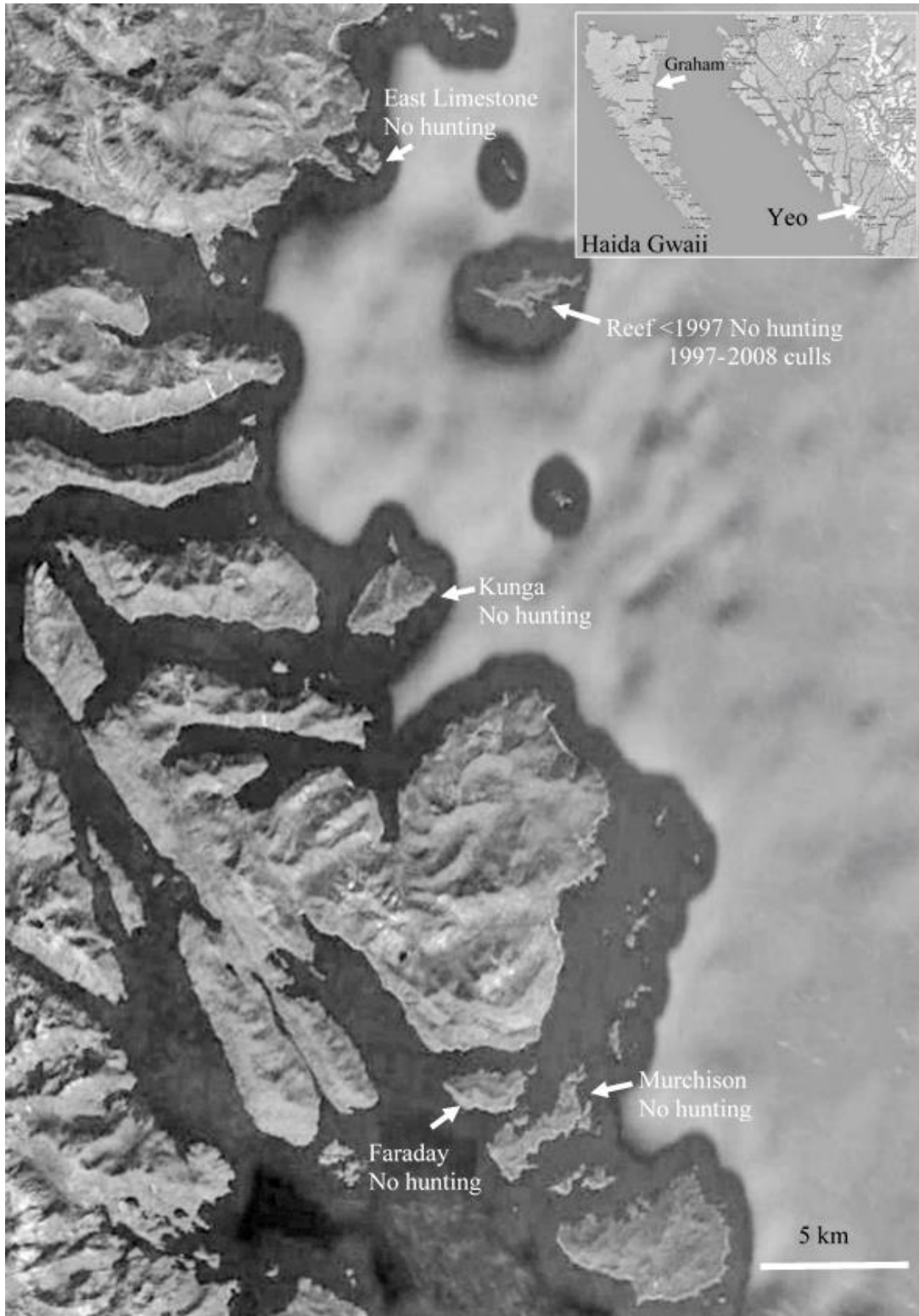
122 Covered by mature forests without human settlements or activities, they were all initially
 123 characterized by open and **species poor** understories (Martin et al. 1995, Stockton et al. 2005,
 124 Martin et al. 2010) representative of severe deer impacts that we documented at the scale of the
 125 archipelago (Martin et al. 2010, Chollet et al. 2015).

126

127 **Table 1.** List of islands included in the study and their key characteristics. Risk hist. = Risk history; Vegetation =
 128 standardized data on vegetation structure and composition; Isotope = collection of bones for the stable isotope study
 129 (section 2.3.4.); Reef <1997 = conditions on Reef prior to experimental culls; Reef >1997 = conditions after the culls
 130 that occurred between 1997 and 2008; Hunting Y = seasonal hunting, Y(cull) = repeated culls; N = no hunting;
 131 Predators Y = present (wolves and possibly cougar and grizzly bear), N = absent; Vegetation Y = standardized
 132 sampling done, N = no standardized sampling; Isotope Y = bone samples collected. * Islands that provided data only
 133 for the study on stable isotopes (section 2.3.4).

Risk hist.	Island	Region	Area	Hunting	Predators	Vegetation	Isotope
No-risk	East Limestone	Haida Gwaii	48 ha	N	N	Y	Y
No-risk	Kunga	Haida Gwaii	395 ha	N	N	Y	Y
No-risk	Faraday*	Haida Gwaii	308 ha	N	N	N	Y
No-risk	Murchison*	Haida Gwaii	425 ha	N	N	N	Y
No-risk	Reef <1997	Haida Gwaii	249 ha	N	N	Y	Y
Risk	Reef >1997	Haida Gwaii	249 ha	Y (cull)	N	Y	Y
Risk	Graham	Haida Gwaii	6361 km ²	Y	N	Y	Y
Risk	Yeo	Coastal BC	95 km ²	Y	Y	Y	Y

134



135

136 **Figure 1.** Map of the study area. Inset = localisation of Haida Gwaii and of Graham and Yeo islands in western British
 137 Columbia, Canada. Main map = location of the islands studied in the central east coast of Haida Gwaii, BC, Canada.
 138 Yeo = hunting and natural predators.

139
 140 Reef Island, initially devoid of hunting, had been subjected to repeated and severe experimental
 141 deer culls between 1997 and 2008 (Chollet et al. 2016). During September 1997 to February 1999,

142 over 80% of the initial deer population was culled. Recent archipelago-wide genetic analyses
143 confirmed that these culls caused a severe population bottleneck and that the current deer
144 population of Reef Island consists of descendants from the handful of animals left after the culls
145 (Burgess et al. 2022a, b, 2023). As local deer survival is typically less than 10 years (JLM et al.
146 unpubl.), all individuals involved in this study (2011-2014) were presumably born after the initial
147 culls. At that time, Reef Island had a deer population density estimated at about 15 deer/km², with
148 an estimated 30 to 40 deer present, and a partially, but dramatically, recovered understory
149 vegetation (Chollet et al. 2016).

150 We added to these three islands a portion of Graham Island (6,361 km²), the largest island of the
151 Haida Gwaii archipelago, where deer have been widespread since the early 20th century
152 (Golumbia et al. 2008) and have been exposed to human hunting ever since. Graham is also home
153 to a population of black bear (*Ursus americanus*) that sometime prey on deer fawns (Mathews and
154 Porter 1988, Ballard et al. 2001). Deer densities on Graham have been estimated to exceed 13
155 deer/km² (Engelstoft 2001, Engelstoft et al. 2008). Its forests are characterized by low vegetation
156 cover in the understory [this study and Chollet et al. (2021b)].

157 Finally, we complemented the sites selected on Haida Gwaii with a site on coastal British
158 Columbia, Yeo Island, situated about 15 km north of Bella Bella, where deer are native and
159 exposed to both human hunting, and predators such as wolves (*Canis lupus*) (Darimont et al.
160 2007), black and brown (*Ursus arctos*) bears, and cougars (*Felis concolor*). Forestry operations
161 occur in parts of Yeo and Graham islands.

162 We thus had access to study sites varying in history of deer presence, in hunting history and in
163 deer exposure to large carnivores. On East Limestone, Kunga and Reef we also had access to 27,
164 23 and 8 marked deer respectively that we ear-tagged and GPS collared during a project that took
165 place from 2011 to 2013.

166 **Is vegetation cover and diversity shaped by predation risk?**

167 To assess the links between the vegetation and predation risk, we compared the understory
168 vegetation among sites with different risk histories (Table 1). We estimated the % cover of plant
169 species in the 0 to 1.5 m vegetation layer directly accessible to deer browsing. We used 3.6 m
170 radius (50 m²) vegetation plots: 20 on Kunga and 10 on East Limestone Islands (long term
171 presence of deer, no-risk); 22 on Reef [long term presence of deer, sampled before the deer culls
172 (no-risk) and after the deer culls (past risk)]; 20 on Graham (long term presence of deer, limited
173 but yearly hunting pressure); and 7 on Yeo (predators present and deer hunting). Details of Graham
174 Island sites can be found in Chollet et al. (2021b). We grouped the plant species data into six
175 groups: young conifers, young deciduous trees, shrubs, ferns, forbs, and other herbaceous plants.
176 To characterize the understory cover of these plant groups in the different study sites we used a

177 **Principal Component Analysis (PCA) on normalised data** in R (R-4.1.0) (Racine 2012)[prcomp
178 function in R (Team 2018)].

179 **Is deer behaviour shaped by predation risk?**

180 **To assess and, when appropriate, quantify and compare deer behaviour in relation to risk history**
181 **we experimentally studied deer Flight Initiation Distance (FID), Distance travelled during flight**
182 **(Dtravel), and deer response to bait and traps among deer populations. We also compared deer**
183 **diel activity pattern in relation to risk.**

184 We obtained behaviour data on islands without risk (Kunga and East Limestone), with no-risk but
185 subject to previous culling (Reef), with hunting risk (Graham) and with risk from both hunting
186 and natural predators (Yeo). To tease apart the roles of risk history and understory food availability
187 in explaining deer behaviour, we carried out a translocation experiment. We moved deer
188 unexposed to hunting and living in a heavily browsed understory (No-risk Kunga Island), to a
189 post-cull island where a much denser understory had resulted from the past experimental reduction
190 in deer density (post-cull Reef Island). We therefore had behaviour data for seven distinct deer
191 samples that varied in the risk contexts to which they were or had been exposed.

192 Finally, we repeatedly observed deer feeding on beached and attached seaweeds in the intertidal
193 area of rocky shores and beaches, particularly at low tides. These observations led us to conclude
194 that beach feeding exposed deer to higher predation risk than feeding within the forest, because of
195 the open environment where escape routes towards cover were limited by the ocean, and where
196 wave sounds hindered the ability to detect danger by ear [for use of shorelines for by foraging
197 wolves see Klein (1995), Darimont and Reimchen (2002), Roffler et al. (2023)]. To test this
198 hypothesis, we used an indirect approach based on stable isotopes to assess if shores were used
199 less at sites with hunting or with hunting and natural predators than no risk sites. For this we
200 collected bones from deer belonging to all the study populations, to which we added samples from
201 Murchison and Faraday Islands, both without hunting (Fig.1 and Table 1).

202 **Is risk affecting Flight Initiation Distance, and Distance travelled?**

203 FID (Flight Initiation Distance) is the distance at which an animal will start moving away at the
204 approach of a potential predator (Ydenberg and Dill 1986). FID has been considered as a key
205 method to disentangle the “economics” of anti-predatory behaviour, as flight occurs where the
206 decreasing value of remaining, and the increasing cost of not fleeing, intercept (Cooper 2008).
207 According to Lima and Dill (Lima and Dill 1990) FID should, under equal resource level, be
208 shorter in safe areas, and longer in risky areas. We therefore used FID as a comparative metric of
209 wariness, using the following protocol: once the observer detected an individual, it was identified
210 by means of its ear tag number or its morphological features. Then the observer walked slowly
211 and calmly (1.8-2 km/hr) towards the deer, avoiding eye contact. The observer stopped walking
212 when the deer changed its initial behaviour and started moving away, but continued observing the

213 animal. We recorded and used as focal variables two distances: distance from the observer at which
214 the deer starts to move away (FID), and Distance travelled (Dtravel) by the deer between its first
215 location (when flight began) and its second location where it stopped moving away and resumed
216 (foraging) activity. We measured these distances by footstep lengths calibrated in meters after
217 each encounter. In the few instances when flight included deer moving out of sight from the
218 observer we estimated a conservative distance travelled by recording the distance to where it
219 disappeared.

220 We ran such FID experiments during four field seasons from spring 2011 to spring 2014, most
221 during 2011-2012. To avoid the confounding **effects of pseudo-replication** and deer habituation to
222 the experiment, we discarded repeated measurements on the same individual and only considered
223 the first FID experiment during each encounter of an individual. We retained 218 FID
224 experiments: 73 from Kunga, 133 from East Limestone, and 12 from Reef islands. We had no FID
225 data for Graham or Yeo islands. We included 19 FID observations for the Kunga deer eight months
226 after their translocation to Reef Island, making up four deer samples for this analysis: Kunga and
227 East Limestone (No-risk), Reef (post-cull), and Kunga-translocated.

228 We analysed the two distance variables for these deer samples with a linear model (lm function in
229 R) to fit the linear regression model with the distance variable (FID or Dtravel, **both log 10**
230 transformed) used as a response variable to assess the effects of risk category. We also analysed,
231 when feasible within deer samples, the effect of deer status (marked/unmarked), sex (M/F) and
232 age class (adult, young) as well as effect of year on FID and Dtravel. We used pairwise
233 comparisons on the linear models with the emmeans package in R which provided t-tests to
234 compare the effects of these variables on our distance variables within deer samples.

235 **Does risk history affect the use of bait and traps, and diel activity?**

236 Neophobia, an adverse reaction to novelty, is an important trait that allows animals to minimize
237 exposure to threats and how to respond to new resources (Greenberg and Mettke-Hofmann 2001,
238 Monestier et al. 2017). We looked at deer use of stations baited with apples and quantified interest
239 in this foreign food by using camera traps (RECONYX PC900) (Le Saout et al. 2015). We also
240 used traps baited with apples to assess the propensity to get trapped as a proxy for exploration
241 behaviour in presence of a foreign object. We baited each bait station daily with 1.5 to 2 apples
242 unless weather impeded fieldwork. For trapping we placed one chopped apple outside the trap
243 entrance, and another one at the furthest end inside the trap near the trigger. Depending on the
244 requirements of each stage of the study, the cameras at bait stations were programmed to acquire
245 from ten to 99 pictures every time the motion sensors were triggered, with a 1s intervals between
246 pictures. A built-in infrared flash with no red glow allowed us to capture images at night or under
247 low light conditions. We recorded: whether or not the deer had been trapped (hence, marked); bait
248 presence at the station (i.e. that it had not been consumed by previous visitors) (yes/no); bait
249 consumption if bait present (yes/no), the time at the beginning and end of a sequence; we also

250 recorded if the deer investigated the ground area where the bait had been before (if bait was absent,
251 yes/no). We set up bait stations monitored with automatic camera for 15 days on East Limestone
252 (no-risk, 4 locations), 12 days on Kunga (no-risk, 4 locations) and 28 days on Reef (post-cull, 8
253 locations). These data were collected between 2011 to 2014, with most visits to bait stations
254 collected in 2011 and 2014.

255 We calculated the rate of visits at bait stations as follows:

256 Rate of visits (visits per day) = (Total visits)/(Nb stations * Nb days stations were monitored)

257 We recorded the tag identity of the marked individuals and identified unmarked individuals using
258 physical features.

259 We used the recorded start time and end time of a feeding sequence to compute the time a deer
260 spent at a bait station when bait was present. We considered time spent at a bait station with bait
261 as a measure of bait friendliness as in Chamaillé-Jammes et al. (2014). We used an analysis of
262 variance on log transformed minutes [aov and emmeans functions in R] to compare “time-spent”
263 among deer samples and between marked and unmarked deer within a deer sample. Time spent at
264 bait stations in the absence of bait was analysed similarly using the camera data collected at
265 stations where bait had been consumed during previous visits. We also analysed trapping rate and
266 the propension of individual deer to be trapped repeatedly.

267 To assess diel activity we used all deer observations obtained from automatic cameras during a
268 study on deer vigilance (Le Saout et al. 2015), and during our investigation on the use of bait
269 stations. We used the time recorded to assign observations to day or night, defining day as the
270 period between civil twilight start (morning) and end (evening) for that date, using Reef Island as
271 the reference locality for civil twilights (time difference with the two other localities is < 20s).
272 (<https://www.timeanddate.com/sun/@6118904?month=5&year=2011>). We compared the
273 proportion of day and night observations among sites with a Pearson’s Chi-squared test (R
274 function `chisq.test`).

275 **Translocation experiment: is deer behaviour shaped by food or fear?**

276 To tease apart the relative effects of resource abundance in the understory and of presence or
277 absence of hunting history, we translocated six adult does from heavily browsed Kunga Island
278 (no-risk). These deer had been trapped repeatedly and had been part of the animals subjected to
279 FID. We translocated them in September 2013 to better vegetated Reef Island (post-cull) where
280 they settled among the local deer born post culls [(Burgess et al. 2022a, 2023) and M.A. Russello
281 pers. com.]. We compared their behaviour on Reef Island with their behaviour on their native
282 island and with the behaviour of the post-cull deer native to Reef.

283 The translocated does had been box-trapped, marked and collared as adults in 2011 and had been
284 recaptured multiple times (5 to 22 times each) (Le Saout et al. 2014, Bonnot et al. 2016). We

285 replaced their GPS collars before translocation. In spring 2014 five translocated animals were still
286 present on Reef. No hunting took place on Reef Island during the eight months the translocated
287 animals spent there from September 2013 to May 2014. In May 2014 we tested them for FID,
288 distance travelled and bait avoidance in the same way they had been tested previously on Kunga.
289 We used six bait stations, set up in spring 2014 and distributed across five locations on Reef Island,
290 keeping them active for fourteen consecutive days.

291 **Stable isotope approach: is deer use of exposed shorelines shaped by food or fear?**

292 To determine the proportion of marine algae in deer diets (a measure of shoreline use) on different
293 islands, we examined stable isotope ratios of Nitrogen and Carbon in deer jaw bones collected
294 from all study sites (Table 1). Because marine plants are enriched in ^{13}C compared to terrestrial
295 plants (Balasse et al. 2005, Richards et al. 2006, Schulting et al. 2008), the isotopic signatures of
296 tissues from coastal herbivores can be used to estimate the relative contribution of terrestrial versus
297 marine plants in their diet (DeNiro and Epstein 1978). We collected bones either from hunting,
298 where hunting occurred, or from the forest floor where deer had died from natural causes. On the
299 larger islands [Kunga, Graham and Yeo] we restricted bone collecting to areas less than 1000 m
300 from the shoreline. We obtained jaw bones from East Limestone (10), Kunga (12), Murchison
301 (10) and Faraday islands (6), from pre-cull Reef Island (7), post-cull Reef Island (6), collected 10
302 years or more after the initial culls (2008 to 2014), Graham Island (23), and from the coastal
303 mainland (17).

304 Stable isotope ratios are expressed in the standard delta (δ) notation, defined as parts per mil (‰)
305 deviation from a standard:

306
$$\delta X = [(R \text{ in sample} / R \text{ in reference}) - 1] * 1000 \text{ (‰)}$$

307 δX gives the deviation between the **samples** isotopic ratio and the ratio obtained from an
308 international standard, “R in sample” is the isotopic ratio considered, in our case $^{13}\text{C}/^{12}\text{C}$ and
309 $^{15}\text{N}/^{14}\text{N}$. For Carbon, “R in reference” refers to the standard Pee Dee Belemnite (PDB) originally
310 derived from a Cretaceous marine fossil, *Belemnitella americana*, from the Peedee Formation in
311 South Carolina, for Nitrogen the accepted standard is atmospheric N_2 .

312 $\delta^{15}\text{N}$ typically get enriched by about 3‰ when going from one trophic level to the next. At the
313 consumer level the whole animal body $\delta^{13}\text{C}$ closely reflects its concentration in the diet but its
314 fractioning varies among tissues. Values of concentration gain vary from + 1.5‰ in muscle, to +
315 5‰ in consumer’s collagen tissue when compared to whole body values (DeNiro and Epstein
316 1978, 1981, Bocherens 1999, Kelly 2000, Bocherens and Drucker 2003, Camin et al. 2016).

317 To account for and assess spatial variability in isotopic composition of the food potentially
318 consumed by deer, we collected samples of 23 dominant plant species in Laskeek Bay (Reef and
319 East Limestone Islands) and on the coastal mainland, grouping them into: coniferous trees (4
320 species), deciduous trees (2 species), deciduous shrubs (9 species), evergreen shrubs (1 species),

321 ferns (5 species), grasses (5 species). We rinsed fresh samples in distilled water to avoid
322 contamination, pre-dried them in the field-camp cabin and fully dried them in the lab at 40°C for
323 24 hours.

324 We collected or had access to samples from one to thirty individuals of 41 seaweed species (19
325 species of brown, 5 green, and 19 red) collected on Reef Island, Louise Island, and Kunghit Island
326 on Haida Gwaii, and on the coastal mainland. We rinsed each sample collected in fresh water
327 immediately after collection and subsequently decarbonated them by a 5s treatment in acid
328 solution at 7.5%. Finally, we rinsed each sample twice in deionised water (Salomon et al. 2008).

329 Bone samples were cleaned and dried according to a protocol defined by late F. Catzefflis based
330 on (DeNiro and Epstein 1981, Bocherens et al. 1988, Bochérens, Hervé et al. 1991) and dry plant
331 and bone samples were ground into powder at a granulometry of less than 0.7mm. Collagen was
332 extracted from bone powder and treated through a standard protocol following the procedure
333 defined by DeNiro and Epstein (1981) slightly modified by Bocherens et al. (1988), and
334 Bocherens, Hervé et al. (1991).

335 We analysed the isotopic composition of the plant and collagen samples on CO₂ and N₂ obtained
336 by sample combustion and analysed on a mass spectrometer.

337 The isotopic signatures of deciduous trees, deciduous shrubs and forbs were extremely similar.
338 Hence, we grouped them into one cluster “DecHerbs”. Ferns and Grasses were also pooled with
339 the deciduous and forb cluster (DecHerbs) on the basis of palatability to deer (Taylor 1956, Pojar
340 1999). On the basis of their differences in isotopic signatures and their lower palatability to deer
341 we retained conifers as a distinct group (mainly represented by Sitka spruce and western hemlock
342 in the understory) (labelled Conifer), as well as salal (the evergreen shrub, labelled EverShrub).
343 Brown, green and red seaweeds had a restricted spread of isotopic signatures and we pooled them
344 into a single group labelled “Seaweed”.

345 We analysed the isotopic ratios obtained for these four groups of plants (Seaweed, DecHerbs,
346 Conifer and EverShrub) and for the bone samples using a Bayesian multiple source mixing model
347 (MixSIAR package in R) (Stock et al. 2018a, b) data to estimate the proportions of source
348 contributions (here terrestrial and marine plants) to a mixture (bones) (Bochérens and Drucker
349 2003).

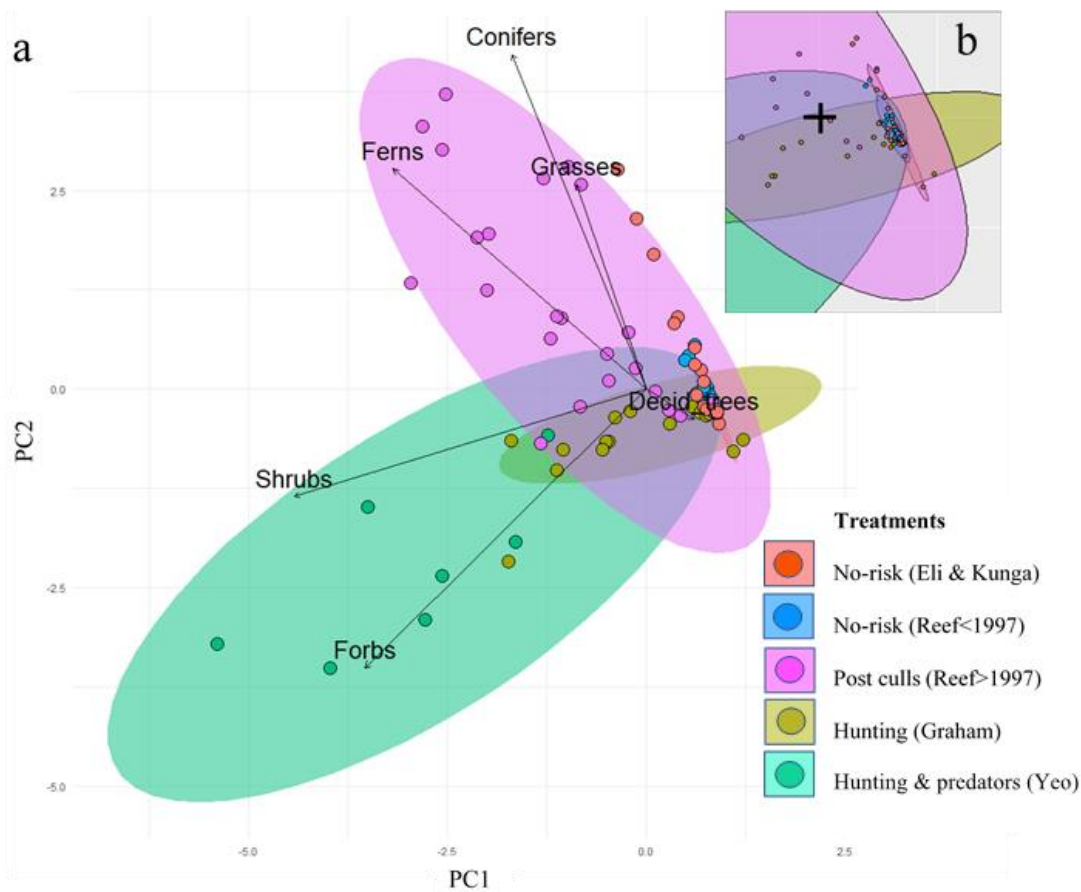
350 **Results**

351 **Understory vegetation varied with risk history**

352 Samples collected in forests with deer, but without a history of hunting (Reef Island prior to culls),
353 Kunga and East Limestone islands), had **low vegetation cover** in all understory plant groups (Figs.
354 2 and 3). Variability among plots was low (large overlap among samples and reduced spread of
355 plots, small ellipses), with only 4 outlying plots with high cover of conifer regeneration (spruce
356 and hemlock) (Figs. 2 and 3). All samples from sites with recent or current hunting, or hunting

357 plus predation (Reef Island post-cull, Graham, and Yeo islands), had a wider spread of plot
 358 coordinates on the **first and second principal component plane** (Fig. 2). Variability among plots in
 359 understory vegetation was highest for the Yeo Island sample (hunting and predators), and the Reef
 360 Island post-cull sample. Yeo Island plots had a high shrub and forb cover (Figs. 2 and 3), while
 361 those on Reef Island post-cull showed high cover in conifers, ferns and grasses and an increase in
 362 forb and shrub cover (Figs. 2 and 3). Among plot variability was lower on Graham Island, exposed
 363 to a long history of yearly low intensity hunting (Fig. 2) and cover of shrubs, forbs and ferns was
 364 higher (Figs. 2 and 3).

365

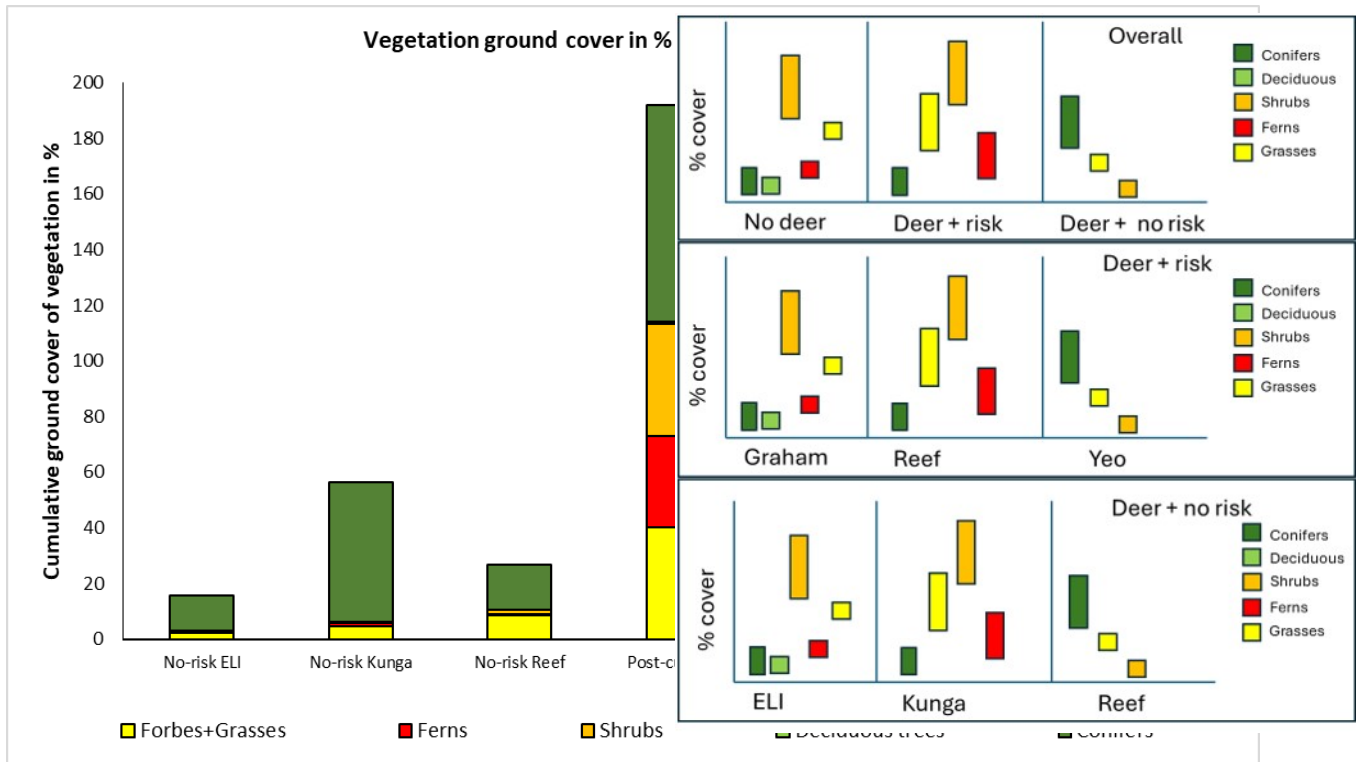


366

367 **Figure 2. a:** Scores of the vegetation plots on the two first components of the Principal Component Analysis on
 368 understory vegetation in relation to the main categories of understory vegetation. Treatments were color coded post
 369 analysis. Treatments are: No hunting represented by [No-risk (East Limestone (Eli) & Kunga)] = East Limestone and
 370 Kunga islands, in red, and [No-risk (Reef<1997)] = Reef island before the cull (1997), in blue; Presence of risk
 371 through hunting with [Past culls (Reef>1997)] = Reef Island after the 1997 to 2008 culls, in pink, and [Hunting
 372 (Graham)] = long history of yearly hunting, in kaki; Presence of risk through regular hunting and presence of deer
 373 natural predators [Hunting + predators (Yeo)] = Yeo island, in green. **b:** zoom around plot center showing the ellipses
 374 corresponding to the [No-risk (Eli & Kunga)] (small red shade) and [No-risk (Reef<1997)] (tiny blue shade). Their
 375 small extent expresses the extreme similarity among vegetation plots in these samples. Note the five outliers from
 376 Kunga in the [No-risk (Eli & Kunga)] sample.

377

378



380
 381
 382 **Figure 3.** Cumulative percent cover recorded in the 0 to 1.5 m strata for the main understory plant categories in the
 383 different sampling units of our treatment categories: absence of predation risk (No-risk East Limestone and Kunga
 384 islands), and No-risk Reef island before the culls (1997); short recent and severe history of hunting (Post-cull Reef
 385 Island (2010); long history of only hunting (Hunting Graham); long history of hunting and of natural predator presence
 386 (Hunting + predators Yeo); No-deer islands = data from three islands on Haida Gwaii (Laskeek Bay) never colonized
 387 by deer (Martin et al., 2010). We lumped herbaceous vegetation into one category.

388
 389
 390
 391 **Flight initiation distance varied with risk category**

392 We collected FID data over several years at East Limestone, Kunga and Reef islands (post-cull):
 393 values among years did not differ significantly ($p = 0.60$). Nor did sex or age have any effect ($p =$
 394 0.30 , $p = 0.62$ respectively). Combining sexes, ages and years, FID varied significantly among
 395 samples ($p < 0.001$; $F = 49.28$ [on 3 predictors, $DF = 233$], residual standard error = 0.24 , $R^2 =$
 396 0.38). All pairwise comparisons differed significantly (Fig. 4a, Table 2). FID was lowest for “No-
 397 risk” samples and highest for the Reef Island post-cull sample. Among the “No-risk” samples,
 398 FID was lowest for the East Limestone Island sample and highest for the Kunga deer translocated
 399 to Reef Island (Fig. 4a, Table 2).

402 **Table 2.** Flight initiation distance (FID) in meter: FID_mean = mean values, FID_sd = standard deviation, and N =
 403 sample size for the different deer categories studied.

404

Deer category	Hunting	FID_mean (m)	FID_sd (m)	N
No-risk ELI (native)	none	6.3	3.7	133
No-risk Kunga (native)	none	11.5	6.8	73
No-risk K on Reef (from Kunga)	none	17.7	12.4	19
Post-cull Reef (native)	culls	28.7	18.0	12

405
 406

407 **Table 3.** Flight initiation distances (FID) in meter and distance travelled after flight (Dtravel) in meter before resuming
 408 activity for the different treatment categories and for marked versus unmarked deer. Events = total number of visits
 409 recorded; Nb. Deer = total number of different deer individuals involved in a category; Total = total number of marked
 410 and unmarked individuals within a treatment. On No-risk Kunga only marked deer were involved in the FID
 411 experiments; the Kunga deer translocated to post-cull Reef (noted No-risk K on Reef) involved, by design, only
 412 marked individuals.

Deer Category	FID	Events	Nb. Deer	Total	Dtravel	Events	Nb. Deer	Total
No-risk East Limestone marked	6.9	78	13	31	6.7	78	13	31
No-risk East Limest. unmarked	5.5	55	18		4.9	56	18	
No-risk Kunga marked	11.5	73	10	10	10.5	72	10	10
No-risk K on Reef marked	17.7	19	5	5	10.1	17	5	5
Post-cull Reef marked	36	5	4	11	41	5	5	10
Post-cull Reef unmarked	23.6	7	7		36.7	6	6	

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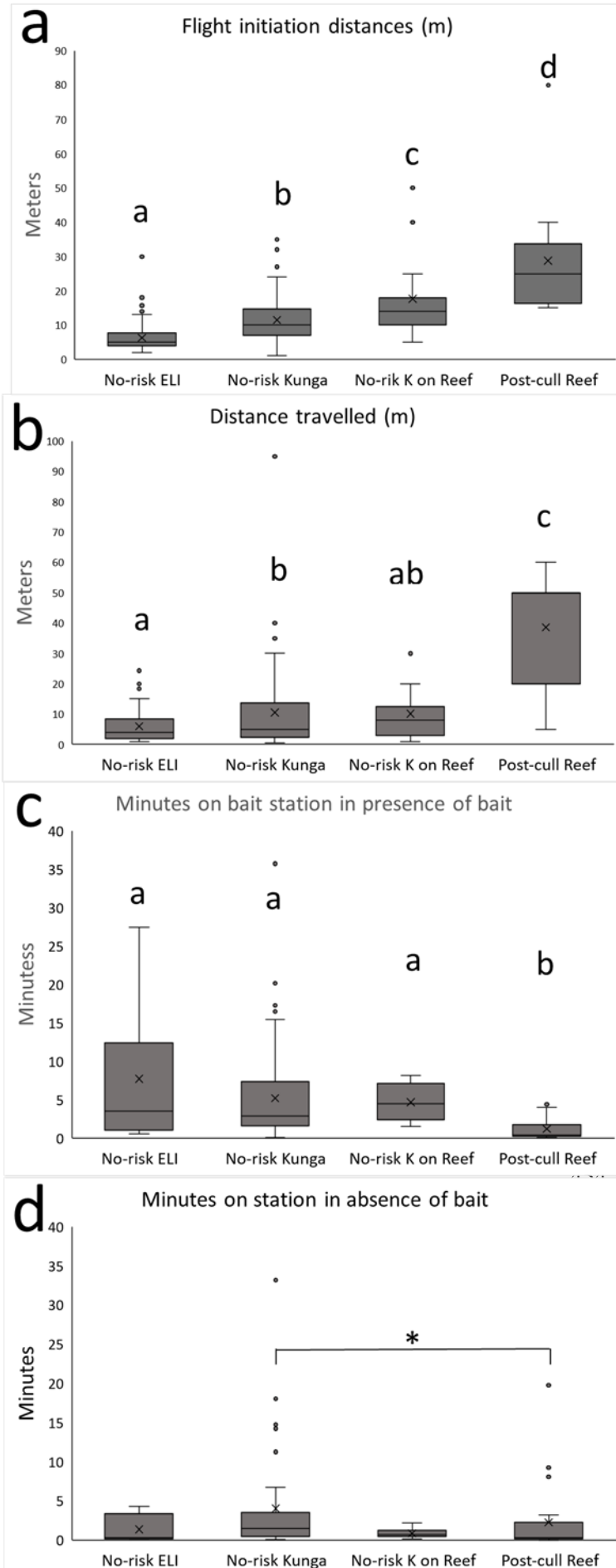


Figure 4. Boxplots on Flight and travel distance and on time spent at bait stations with bait present or absent. No-risk ELI = East Limestone deer, No-risk Kunga = Kunga deer on Kunga Island, **No-risk K on Reef** = Kunga deer after translocation to Reef Island, Post-cull Reef = Reef Island deer born into a population that survived the 1997 to 2008 culls. X in box = mean value, solid line in box = median value. Letters indicate results of post-hoc tests on the linear regression model on FID (log₁₀ transformed) as a function of deer categories.

a. Flight Initiation Distance (FID). All pairwise comparisons were significant: most had p-values < 0.001 except No-risk Kunga – No-risk K on Past culls Reef (p = 0.02) and No-risk K on Post-cull Reef – Post-cull Reef (p = 0.03).

b. Distance travelled after flight initiation (Dtravel). For Post-cull Reef Median = 50 m. All pairwise comparisons were significant except for No-risk (ELI – No-risk K on Reef (p = 0.12) and No-risk Kunga – No-risk K on Post-cull Reef (p = 0.96). For the pairwise comparisons that were significant all p-values were < 0.001 except No risk ELI – No-risk Kunga (p = 0.01).

c. Time spent at bait stations with bait present. All comparisons among No risk categories were non-significant (p-values > 0.80). For the pairwise comparisons that were significant, all p-values were < 0.05.


d. Time spent at bait stations by deer when bait was missing (consumed since station was re-provisioned). All comparisons among No risk categories were non-significant (p-values > 0.50, except for No-risk Kunga – No-risk K on Post-cull Reef (p = 0.37). Only for the No-risk Kunga – Post-cull Reef comparison did Kunga deer spent significantly more time at stations with no bait left than did deer native to Post-cull Reef (p = 0.04).

457 **Table 4.** Distance Travelled (Dtravel) after flight initiation: Dtravel_mean = mean values, Dtravel_sd = standard
 458 deviation and N = sample size for the different deer categories studied.

Deer category	Hunting	Dtravel_mean (m)	Dtravel_sd (m)	N
No-risk East Limestone (native)	none	6.0	5.8	133
No-risk Kunga (native)	none	10.5	13.7	72
No-risk K on Reef (from Kunga)	none	10.1	8.9	17
Post-cull Reef (native)	cull	38.6	18.4	11

459
 460 Only the samples from East Limestone and Reef Island (post-cull) included marked deer. There
 461 was a marginal trend for longer FID in “No-risk” East Limestone marked deer (t-ratio = 1.85, p =
 462 0.07) (6.9 versus 5.5 m). There was no statistically significant difference in FID between marked
 463 and unmarked deer in “post-cull Reef” (t-ratio = 0.95, p = 0.34) (36 versus 23.6 m) (Table 3).

464 **Distance travelled after flight varied with risk category**

465 Dtravel varied among samples with a significant contrast between the “No-risk” and the “post-
 466 cull” samples: $p < 0.001$; F-statistic = 18.04 (on 3 predictors, DF = 229), Residual standard error
 467 = 0.42, $R^2 = 0.18$. (see Fig. 4b). Dtravel observed on East Limestone Island (No-risk) was
 468 significantly shorter than on Kunga Island (No-risk) ($p = 0.01$), and was much shorter Reef Island
 469 post-cull (Fig. 4b) ($p < 0.001$). Dtravel for Kunga Island deer after translocation to Reef was
 470 similar to that observed on Kunga itself ($p = 0.96$), and to that on East Limestone Island (No-risk)
 471 ($p = 0.12$), but much shorter than that observed for deer native to Reef (“post-cull Reef”) ($p <$
 472 0.001) (Fig. 4b, Table 4). 


473 Only the samples from “No-risk East Limestone” and “post-cull Reef” involved marked and
 474 unmarked deer. For “No-risk East Limestone”, Dtravel (log10 transformed) was significantly
 475 higher for marked deer (6.7 m versus 4.9, t-ratio = 2.20, $p = 0.03$). There was no significant
 476 difference in Dflight between marked and unmarked deer in “post-cull Reef” deer (41m versus
 477 36.7, t-ratio = 0.17, $p = 0.86$) (Table 3).

478 **Attraction versus aversion varied with risk category**

479 **Response to bait** – Risk category had a significant effect on time spent at bait stations when bait
 480 was present (log10 transformed): $p = 0.007$, F-statistic = 4.29 (on 3 predictor, DF=79); Residual
 481 standard error = 0.63, $R^2 = 0.14$ (Fig. 4c).

482 When bait was present, deer on islands with no risk, and deer from Kunga translocated to Reef,
 483 spent significantly more time at bait stations than the native Reef Island deer post-cull (Fig. 4c).
 484 The rate of visits at bait stations on islands with no risk history (East Limestone and Kunga)
 485 averaged 0.97 visits/day/station, against 0.20 (five folds lower) for the deer native to Reef post-

486 cull (Table 5.). Frequency of bait consumption per visit exceeded 75% on Kunga and East
487 Limestone (no-risk) (Table 5.) and was 20% for animals native to after the culls on Reef. In the
488 presence of bait, marked deer on East Limestone and Kunga consumed bait in over 80% of the
489 events whereas unmarked deer consumed bait in 61% and 50% respectively of the visits. This is
490 in striking contrast to native deer on Reef post-cull, where only three of the nine native deer
491 trapped and marked post-cull were seen consuming bait at bait stations. Bait remained untouched
492 by the 14 remaining (unmarked) Reef post-cull deer. Unmarked individuals, either briefly
493 investigated the bait (visually and/or olfactory) without consuming it, or ignored it altogether,
494 spending <30 s on site. When comparing median values, less affected by outliers, length of median
495 time spent on a station exceeded 3 min for deer with no hunting history and was less than 30 s for
496 post-cull deer on Reef.

497 As most deer on East Limestone (no-risk) were marked, all visits to a bait station with bait involved
498 marked deer. The deer translocated from Kunga to Reef islands were also all marked. On “No-
499 risk Kunga” and “post-cull Reef” our samples included both marked and unmarked deer. For “No-
500 risk Kunga” deer, seven marked and 21 unmarked deer consumed bait. Forty-two out of 56 visits
501 led to bait consumption. Time spent at a station was significantly higher in marked deer (7.8 min
502 on average) than in unmarked deer (5.6 minutes on average) (t-ratio of contrast estimate = 4.64, p
503 < 0.001). Only unmarked animals visited a bait station without eating the bait, and their mean visit
504 length was less than a minute on average (0.9 min). We observed a similar pattern of longer visits
505 by marked individuals among deer native to Reef post-cull (t ratio of contrast estimate = 3.00, p =
506 0.004). Bait was exclusively consumed by the 4 marked individuals (mean visit length 3.0
507 minutes). The 14 unmarked deer native to Reef Island post-cull visited bait stations on average for
508 0,4 minutes and never consumed any bait. 

509 In absence of bait because of previous consumption, mean visit length by deer at a station were
510 shorter (Fig. 4d versus Fig. 4c). Sample still had a significant effect on length of visit at a station
511 (log10 transformed): p = 0.05, F-statistic = 2.72 (on 3 predictor, DF=84), Residual standard error
512 = 0.64, R^2 = 0.09 (Table 5, Fig. 4d), but this effect was unrelated to risk history. Visits were longer
513 on average on Kunga essentially through the effect of outliers spending long periods at sites that
514 had bait recently (Fig. 4d). Whatever the sample, time spent at a station never exceeded 4 minutes
515 (Table 5). Median values equaled 36 s or less, with the exception of Kunga deer (median = 84 s).
516 When time was spent at a station without bait, it was spent sniffing at the location where bait had
517 been present.

518 On Reef, post-cull marked native deer spent significantly longer at stations without bait than
519 unmarked native deer (means of 2.8min versus 0.13 respectively) (t-ratio of contrast estimate =
520 4.0, p < 0.001). On “No-risk Kunga”, there was no significant difference between marked and
521 unmarked deer in time spent at stations (84 versus 90 s) (t-ratio of contrast estimate = 1.42, p =
522 0.16).

523 **Table 5.** Deer visits and use of bait stations with bait present in relation to island/hunting history category. Stations
 524 (days) = number of stations set up and number of days each was active; Nb. Visits = total number of photographic
 525 sequences a deer was captured on camera at a bait station; With bait = number of visits when bait was present; Eat =
 526 number of visits in which bait was consumed when present; Mean length (median) min = mean value of a visit duration
 527 in minutes and the corresponding median value; Without bait = number of visits when bait was absent (= consumed
 528 during visits that followed the re-provisioning of the station; Nb. deer = total number of different individuals involved
 529 in the experiment; Marked = number of marked individuals among the total number of different individuals involved.

Island/hunting history	Stations (days)	Nb visits	With bait	Eat	Mean length (median) min	No bait	Mean length (median) min	Nb. deer	Marked
No-risk East Limestone	4 (15)	15	8	7	7,7 (3,5)	6	1,4 (0,3)	3	3
No-risk Kunga	4 (12)	90	56	42	5,2 (2,9)	35	4,0 (1,4)	28	7
No-risk K on Reef	5 (16)	31	6	6	4,7 (4,5)	25	0,9 (0,6)	5	5
Post-cull Reef (2011)	8 (35)	9	6	0	1,2 (0,4)	25	2,25 (0,3)	18	4
Post-cull Reef (2014)	5 (16)	35	13	4					

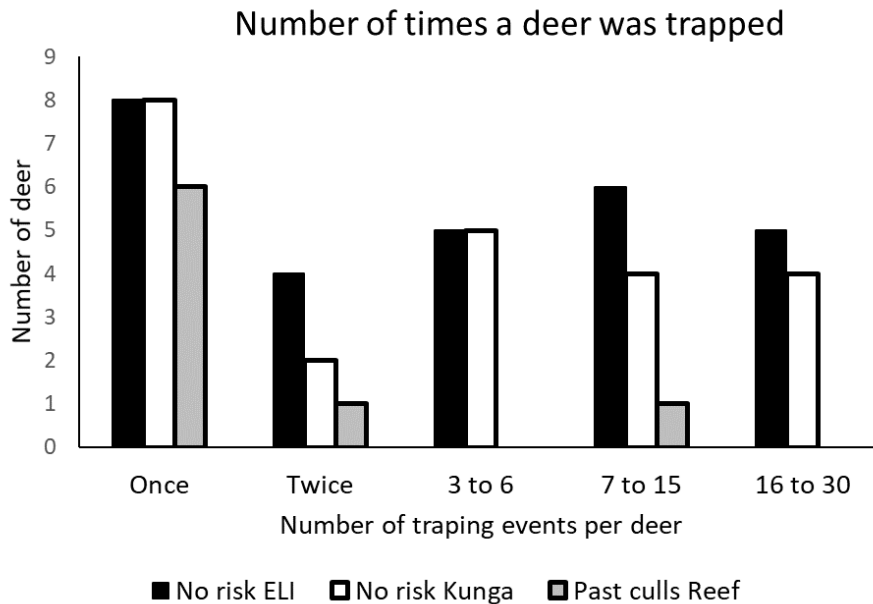
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532 **Response to traps** – On Reef Island post-cull the capture rate was 90% lower than on East
 533 Limestone and 80% lower than on Kunga (Table 6). Only one third as many animals were captured
 534 on Reef Island post-cull as on the two “No-risk” islands, despite a larger number of capture days
 535 in the former than on the latter (five-fold versus six-fold respectively) (Table 6). The difference in
 536 recapture rates is even more dramatic: 45% and 33% of the deer captured on “No-risk” East
 537 Limestone and Kunga islands were recaptured three times or more (up to 30 times for some
 538 individuals on “No-risk East Limestone”) (Fig. 5). On East Limestone Island, 19 out of 27, and
 539 on Kunga, 15 out of 23 deer, were captured more than once. Even more, 13 of the deer captured
 540 on each island were recaptured three times or more. In contrast, among the deer native to post-cull
 541 Reef Island, only two of nine deer were captured more than once: one twice, and the other
 542 individual repeatedly (Fig. 5).

543 **Table 6.** Summary of box-trap capture data of animals on their native island for all sessions. Total captures include
 544 new deer (i.e. first time captured and marked) and recaptures of the marked individuals. ELI = East Limestone Island.

Island	Years	Effort boxes * days	Nb. deer captured	captures + recaptures	Trapping success deer/box*day/year
No-risk ELI	2011- 2013	494	27	207	0.42±0.21
No-risk Kunga	2011- 2013	591	23	150	0.22±0.16
Post-cull Reef	2011- 2013	664	8	23	0.04±0.02

545



546

547 **Figure 5.** Summary of the capture and recapture histories for the deer native to the three sites and two categories of
 548 hunting histories. ELI = East Limestone Island.

549 **Diel activity pattern varied with risk category**

550 Between 2011 and 2014 we recorded 762 (Table 7) picture sequences of deer on automatic
 551 cameras. We used the time recorded on the first picture in a sequence to assign the sequence to
 552 day time or night time. These pictures were taken in spring and early summer, hence a period of
 553 the year of long days and short nights. After correcting for differences in duration of day and
 554 night, deer were more active by day than by night (Fig. 6). There was no significant difference in
 555 daily activity patterns between East Limestone and Kunga islands (no-risk) (Chi-squared = 0.29,
 556 df = 1, p = 0.59) but deer native to post-cull Reef were more nocturnal than either (Chi-squared =
 557 10.38, df = 1, p = 0.005). There was no significant difference between marked and unmarked deer
 558 in diel activity either on Reef Island post-cull or on Kunga Island (Chi-squared = 0.05, df = 1, p =
 559 0.81).

560

561 **Table 7.** Distribution of deer observations by automatic cameras across sites and their assignment to day or night
 562 with day defined as the period between start of civil-twilight in the morning and end of civil-twilight in the evening.
 563 Night was defined as the period between civil twilight end and civil twilight start the following day.

Island/hunting history	N	day	night	% at night
No-risk East Limestone	289	256	33	11%
No-risk Kunga	358	311	47	13%
Post-cull Reef	115	88	27	24%

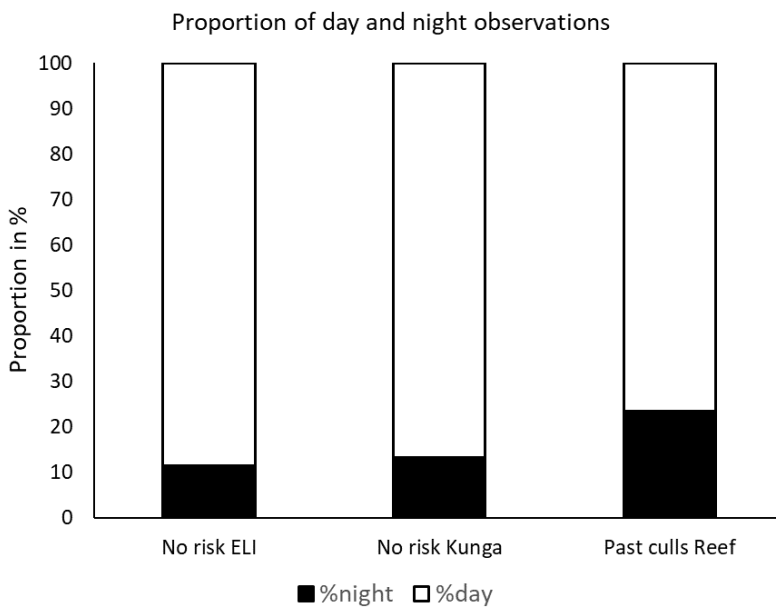


Figure 6. Temporal distribution between day and night of deer caught on automatic cameras near bait stations on islands in the absence of predation risk No-risk ELI (East Limestone) and No-risk Kunga (Kunga) and in presence of past risk (Post-cull Reef)

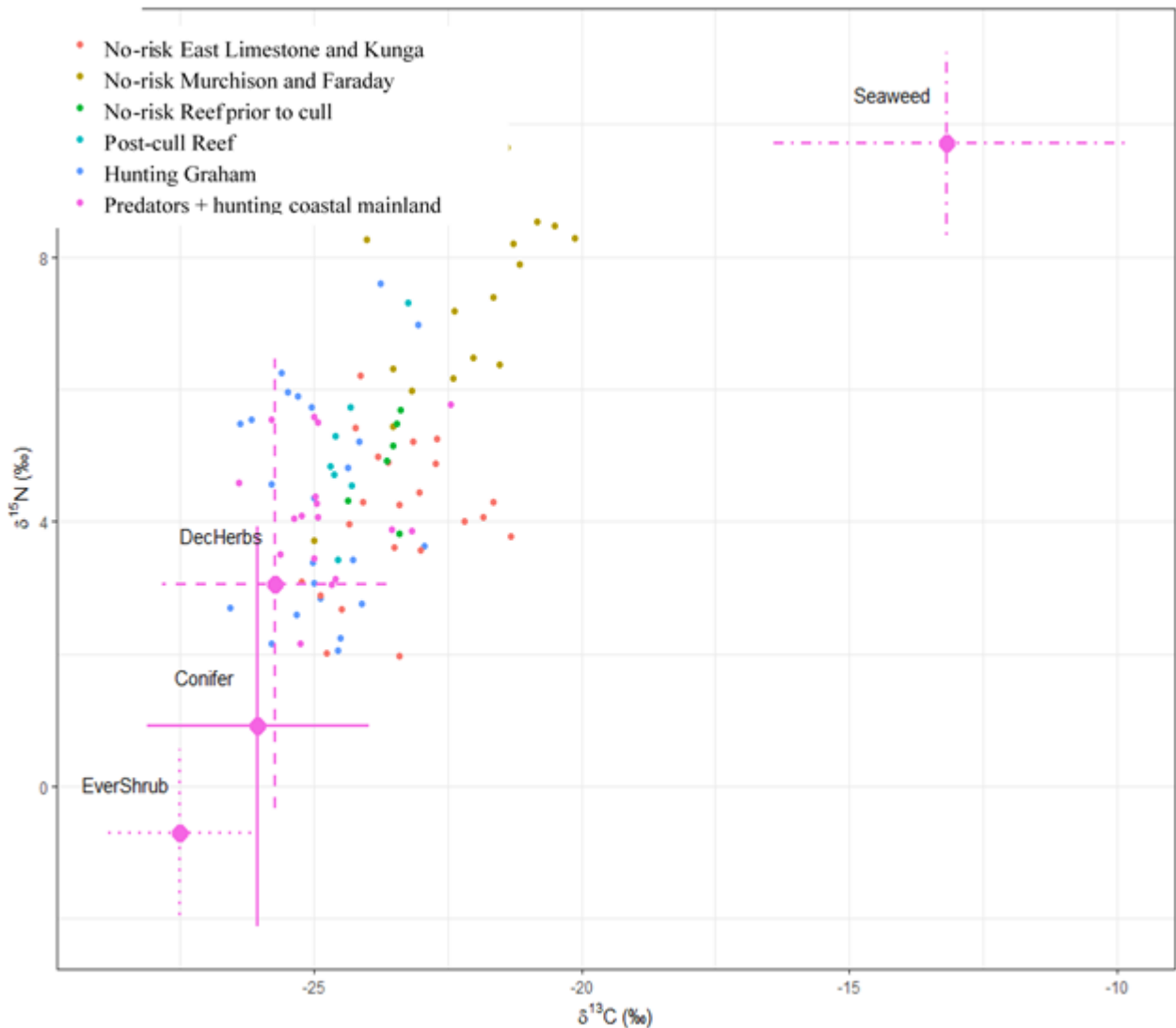
580

581 **Stable isotopes as indicator of the propensity to use exposed shorelines**

582 The Gelman-Rubin Diagnostic run after the MixSIAR model indicated that only 2 of the 140
 583 variables exceed marginally the threshold value of 1.01. The Geweke diagnostic, a standard z-
 584 score to assess if less than 5% of the 140 variables were outside the +/-1.96 values in each chain
 585 indicated 3%, 5% and 1% for chains 1, 2 and 3 respectively. These diagnostic and tests confirmed
 586 that our Markov Chain Monte Carlo (MCM) chains converged.

587 The isospace plot (Fig. 7) shows a spread of the isotopic signature in deer bone samples relative
 588 to the proportion of seaweed and terrestrial plants in deer diet. **The samples with the highest**
 589 **proportion of seaweed-isotopes in jawbone collagen were from the “No-risk” islands Murchison**
 590 **and Faraday.** Those with the least proportion of seaweed isotopes in jawbone collagen were from
 591 the islands with risk, “Hunting Graham” and “Predators + hunting coastal mainland”. The plot
 592 also indicates a ranking of the terrestrial sources in deer diet. The DecHerbs cluster (Deciduous
 593 trees and shrubs, ferns, forbs and grasses) (Fig. 7) made up for the highest proportion. Conifers
 594 came next and the evergreen shrub salal (*Gaultheria shallon*) last.

595 The MixSIAR posterior plots (Fig. 8) illustrate the proportion of the four plant categories retained
 596 in deer diet in the different deer categories. The proportion of seaweed isotopes in bone collagen
 597 reached 31% on “No-risk” Murchison and Faraday, and was 19.2% in “No-risk” sites in Laskeek
 598 Bay (East Limestone, Kunga and Reef prior to the cull). It dropped to 8 and 9.6% respectively in
 599 sites with hunting (“Hunting Graham”) or with predators and hunting (“Predators + hunting coastal
 600 mainland”). It was intermediate (13.5%) in the post-cull Reef deer born after the cull.

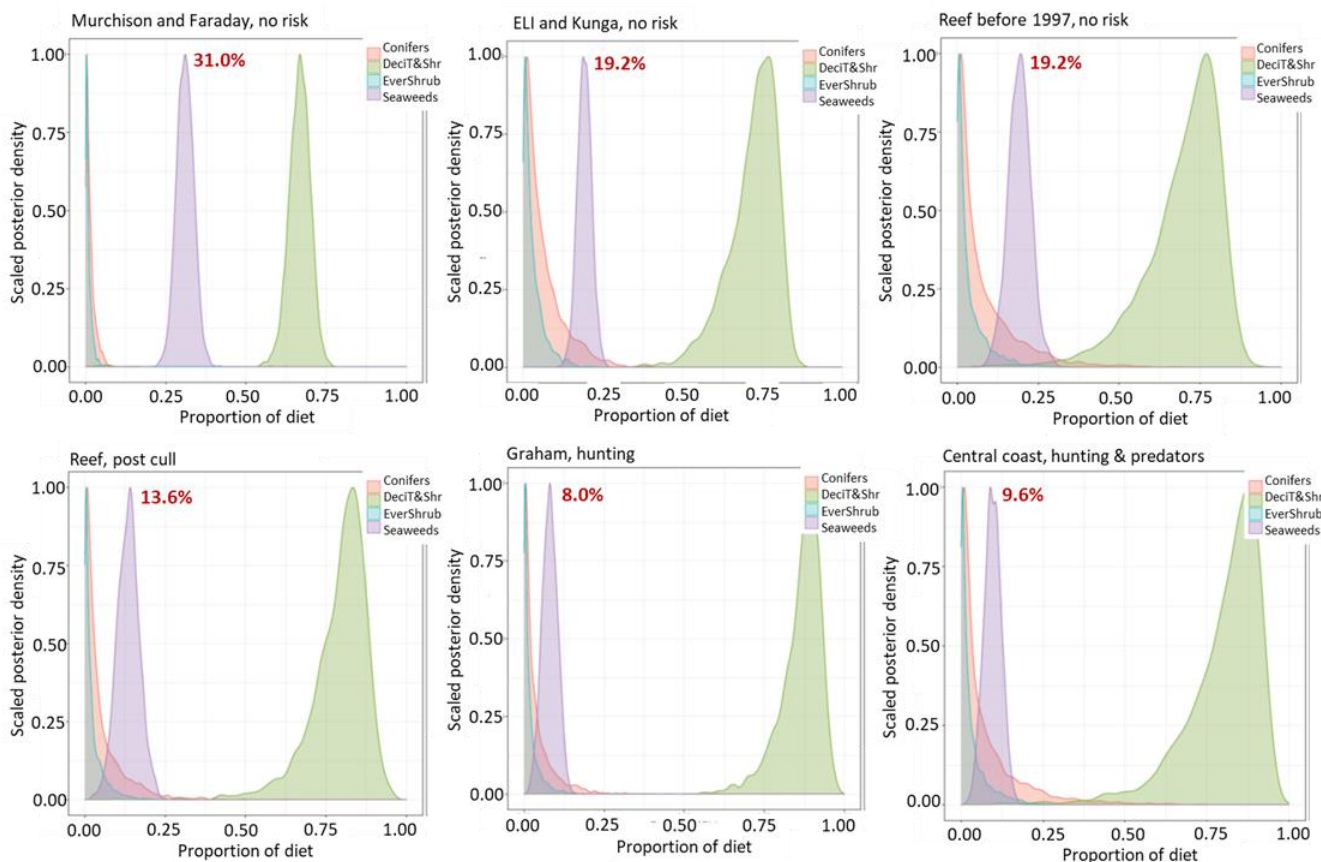


601
 602 **Figure 7.** Isospace plots of deer and plants Mixture data (deer) by risk category and islands. No-risk Murchison and
 603 Faraday = No-risk samples from Murchison (N=10 bone samples) and Faraday (N=6); No-risk samples from East
 604 Limestone (N=10) and Kunga (N=12); No-risk samples from Reef Island prior to the culls (N=7); Post-cull Reef =
 605 samples from Reef Island after the culls (N=6); Hunting Graham = samples from Graham Island (N=23); Predators +
 606 hunting = samples from the coastal mainland (N=17). Source data are by risk/island categories and have been adjusted
 607 by discrimination means and SDs. Terrestrial plants, EverShrub = evergreen shrub, Conifer, DecHerbs = deciduous
 608 trees and shrubs, forbs, ferns and grasses; Seaweed (40 species). Error bars indicate 1 SD, the combined
 609 source+discrimination SD calculated under assumption of independence as: $\sqrt{\sigma_{source}^2 + \sigma_{discr}^2}$

610
 611 Bayesian credible intervals at 95% of seaweed proportions (Fig. 8) confirm the contrast in the
 612 proportion of seaweeds between the samples from no-risk situations (Murchison, Faraday, East
 613 Limestone, Kunga and Reef before the cull), and the two sites with hunting and/or predators
 614 (Graham and coastal mainland). They also illustrate the intermediate position of the post-cull Reef
 615 sample.

616 In all situation the isotope data suggests that deer diets were dominated by the terrestrial plants
 617 group that included deciduous trees and shrubs, ferns, forbs and grasses. Its proportion varied from

618 67% on “No-risk” Murchison and Faraday samples to 88% on “Hunting Graham” and in
 619 “Predators + hunting coastal mainland” samples. Conifers and evergreen shrubs stand out as a
 620 minor component of diet, despite their prevalence in the vegetation profiles of some of our study
 621 sites (see Fig. 3). The proportion of seaweeds in the “No-risk” Murchison and Faraday samples
 622 are remarkably high.



623
 624 **Figure 8.** MixSiar posterior plots of overall deer diet in relation to predation risk. N = number of bone samples per
 625 site. No hunting history, four sites: Murchison (N=10), Faraday (N=6), East Limestone (ELI) (N= 10), Kunga (N=12)
 626 and Reef before the culls (N=7)) (see map in Fig. 1); Reef post-cull = samples from Reef Islands collected after the
 627 culls (2008-2014) (N=6); Yearly relatively low intensity hunting without marked effect on high deer density
 628 (Engelstoft, 2001) Graham Island coastal area (N=23); Coastal mainland, hunting and predators present (N=17). All
 629 profiles are at the same scale. Murchison and Faraday are two islands with vegetation poor understories (see text and
 630 Martin, Gaston & Hitier (1995)). The peaks of the shaded areas represent the median estimate of the proportion of the
 631 different sources in deer diet. Conifers (Sitka spruce, western hemlock and western and yellow redcedar) (pink
 632 shading) (4 species, 13 samples), evergreen shrubs (light blue shading) (1 species (salal), 4 samples), green shading
 633 = deciduous trees and shrubs, forbs, ferns and grasses) (27 species, 86 samples). Seaweeds (purple shading) (40
 634 species, 237 samples). Red figures indicate median percent value).

635
 636
 637

639 How did understory vegetation cover and dynamics change with predation risk?

640 Our results illustrate the well-established effect that large herbivores have on vegetation in the
641 absence of predation risk (Côté et al. 2004, Martin et al. 2010a, 2011, Cardinal et al. 2012a, b,
642 Chollet et al. 2021b). They also illustrate the relationship between the impact of herbivory on
643 the vegetation and the risk history a deer population is exposed to (Callan et al. 2013, Waller
644 and Reo 2018). The severe culls on one of our sites (Reef Island) resulted, over a decade, in an
645 upsurge of understory vegetation and of regenerating conifers in an initially heavily-browsed
646 open understory (Figs. 2 and 3). The increase in **beta diversity (spread of plots, Fig. 2)** in the
647 post-cull Reef Island sample illustrates the response of vegetation cover and composition
648 (Chollet et al. 2016). However, there are intrinsic differences between this recovering
649 understory vegetation and the understory vegetation we sampled on Yeo Island where deer
650 have always been subject to their natural predators and hunters. On post-cull Reef Island, we
651 interpret the high proportion of regenerating conifers in the understory as the legacy of heavily
652 browsed pre-cull patches of dwarfed spruce and hemlock and their dramatic release after the
653 culls (Chollet et al. 2016). By contrast, the vegetation profile we observed in the presence of
654 large carnivores and hunting on Yeo, close to the coastal mainland, is remarkably similar to
655 the understory vegetation profile we observed on three islands in Laskeek Bay where we found
656 no evidence of deer [see details in (Stockton et al. 2005, Martin et al. 2010)]. Their understories
657 lack the significant conifer cover observed on post-cull Reef and are dominated by shrubs,
658 forbs and grasses (Fig. 3).

659 On Graham Island, where hunting is of low intensity and restricted to sites most accessible to
660 the small local population of hunters (Martin and Baltzinger 2002), hunting had a smaller effect
661 in mitigating deer over-browsing, resulting in a beta diversity (spread of plots, Fig. 2) and
662 proportions of different vegetation components (Fig. 3) intermediate between Yeo Island and
663 islands without predation (Engelstoft 2001, Engelstoft et al. 2008, Chollet et al. 2021b)]. This
664 suggests an interplay among the presence of risk, its intensity (hunting versus culling), its
665 nature (humans versus natural predators) and its duration in the interaction between deer and
666 their habitat.

667 Does lack of risk favor bold deer, and does presence of risk favor shy deer?

668 Our comparison of anti-predator behaviours between populations that had not been exposed to
669 hunting (East Limestone and Kunga Islands), with a population on Reef Island that had been
670 subject, over a decade prior to the study, to a severe population reduction through experimental
671 culling, revealed clear behavioural contrasts among these populations. Deer exposed to
672 different risk history responded differently to cues mimicking predation risk. They also differed
673 in their response to foreign food at bait stations, and to foreign food associated with an

674 unfamiliar object. The significantly shorter flight initiation distances, and much shorter
675 distances travelled, observed in non-hunted populations were retained by the Kunga deer
676 translocated to Reef, even though they showed slightly longer flight initiation distances in their
677 novel environment. This was in stark contrast with the wary behaviour of the deer native to
678 post-cull Reef, even more so as the flight and travel distances we recorded for the post-cull
679 Reef deer were likely conservative: they often fled before being seen (flight diagnosed by ear)
680 and often were still running when they disappeared from view. This suggests that we
681 underestimated distances travelled and that actual values on Reef may have been closer to
682 values reported in black-tailed deer populations on the mainland (FID ~ 60 to 70 m and Dtravel
683 ~ 70 to 120 m), where deer can be subject to hunting, poaching and carnivores (Stankowich
684 and Coss 2006, 2007, Stankowich 2008). Our field notes also indicated that animals native to
685 post-cull Reef ran or trotted away, whereas deer from Kunga and East Limestone (no-risk
686 islands) usually performed a slow walk to a nearby location where they resumed their activities.

687 Among the “No-risk” deer categories, FID and distances travelled were significantly shorter
688 on East Limestone than on Kunga. This may reflect some habituation to human presence
689 (Rodriguez-Prieto et al. 2009, Schuttler et al. 2017) on East Limestone Island where people are
690 present daily for two to three months each summer for ecological monitoring.

691 Despite minor differences in flight distances or time spent at bait, marked and unmarked deer
692 of the same island had comparable behavioural profiles. In particular, marked and unmarked,
693 deer from post-cull Reef Island were predominantly shy, as is illustrated by their poor trapping
694 rate, even when we take into account that deer densities on post-cull Reef were still at only
695 50% of their pre-cull density (~30 deer/km², i.e. >80 deer present) (Chollet 2012). Only two
696 (25%) of the eight deer captured on post-cull Reef were ever re-captured, and only one was
697 recaptured repeatedly, mostly as a yearling and a juvenile, before becoming trap shy the
698 following years while still captured on cameras. This contrasts with deer unexposed to risk on
699 East Limestone and Kunga, where over half of the deer trapped were re-trapped at least 3 times,
700 often much more often over the years (Fig. 8).

701 The presence on Kunga and Reef (post-cull) of a fraction of animals never trapped, but captured
702 on automatic cameras, indicated within population variation in trap-shyness of individuals (Sih
703 et al. 2012). Observed differences in behaviour between deer native to East Limestone and
704 Kunga Islands (no-risk) and those native to Reef Island post-cull, suggest that culling shifted
705 the dominant behaviour profiles of Reef Island deer towards the shy-end of a boldness-shyness
706 continuum.

707 Food limitation and nutritional status have also been proposed as mechanisms explaining
708 individual attenuation in antipredator behaviours such as flight response to an approaching
709 threat (Stankowich 2008, Gaynor et al. 2019). Such individual behavioural-adjustments,
710 motivated by food shortage, should be lifted once food supply improves. This could explain

711 the higher values of FID and Dtravel observed for Reef Island deer post-cull, as well as the less
712 wary behaviour of deer living in heavily browsed understories. However, we did not observe
713 any effect of food availability in the deer we translocated from Kunga to Reef. These animals
714 did not change their behaviour despite a large increase in their available food supply. Our
715 observations better fit the hypothesis that less wary behaviour was enabled by the absence of
716 risk. Less wary behaviour could progressively favour individuals investing more in foraging
717 and less in costly anti-predator behaviours. Less costly anti-predator behaviours such as routine
718 vigilance would be retained and not counter-selected as observed for the populations of East
719 Limestone and Kunga where routine vigilance persisted in an otherwise tame population.
720 (Chamaillé-Jammes et al. 2014, Le Saout et al. 2015).

721 In non-hunted populations we suggest that the need to adjust to an increasingly depleted
722 understory explains a progressive shift towards bolder behaviour in the population. On Reef,
723 we suppose that severe culling may have resulted in a strong selection of the wariest individuals
724 best equipped to elude hunters (Terry Husband pers.com). From an initial population not
725 hunted and living in a heavily browsed habitat (Martin et al. 1995, Chollet et al. 2016), those
726 animals that survived were shyer and with a tendency to be more active at night than the mean
727 for the pre-cull population. The higher proportion of night time activity in post-cull Reef deer
728 is consistent with the predicted shift towards more nocturnality in response to increased hunting
729 (Altendorf et al. 2001, Bonnot et al. 2020, Palmer et al. 2022). The overall predominance of
730 diurnal activity in the “No-risk” East Limestone and Kunga deer populations has been shown
731 to contrast with the behavioural patterns commonly observed elsewhere in hunted populations
732 (Bonnot et al. 2016). But a higher proportion daytime activity in non-hunted deer populations
733 on Haida Gwaii may also be interpreted as responding to the need for increased foraging time
734 in intensively browsed understories.

735 **Does attenuation of anti-predator behaviour favour deer density?**

736 Stockton et al. (2005) showed that, even on islands with the most severe deer impact on the
737 understory within our study area, deer densities remained high, exceeding 20 deer per km².
738 However, Le Saout et al. (2014) showed that less than 5% of the resources needed by these
739 dense deer populations were provided by the rare standing biomass. In summer the bulk of deer
740 forage was provided equally by a cryptic flux of growing vegetation and by canopy fall in the
741 form of foliage shed by wind. In winter canopy fall was the dominant source of forage
742 supplemented by seaweed available on the shorelines (Le Saout et al. 2014). But these
743 resources, while sufficient, occurred in small spatially-dispersed cryptic items in contrast with
744 the volume and distribution of palatable foliage and plant material in lush understories on
745 islands without deer (Martin et al. 2010) or with limited deer impact as on Yeo (Fig. 3).
746 Spatially dispersed food occurring as small items, could favour behaviour that increases
747 foraging time at the expense of time devoted to costly anti-predator behaviours (Réale et al.

748 2010, Sih et al. 2012) such as flight or dynamic adjustments of spatial distribution in response
749 to perceived distribution of risk (Williams et al. 2008, Gaynor et al. 2019). This could favour
750 a progressive decrease in understory vegetation, as was actually shown on these islands by
751 (Chollet et al. 2015) between 1989 and 2009.

752 **Did (the absence of) risk affect the use of exposed habitats?**

753 Cursorial predator hunting by sight should increase the reluctance in their prey to use open
754 habitats (Bonnot et al. 2017). In addition, the lush understory on Yeo Island, similar in cover
755 and diversity to the understory of islands never colonized by deer, or the lush understory on
756 post-cull Reef Island, may reduce the incentive to use the shoreline in these deer. The data on
757 understory vegetation (Fig. 3), also suggest a potential for a negative relationship between the
758 amount of vegetation in the understory and the proportion of seaweed signature in deer bone
759 collagen. The very high seaweed isotopic signature on “No-risk” Murchison and Faraday,
760 could, for instance, be interpreted as the consequences of a lack of understory forage on these
761 two islands. The vegetation data collected in 20 plots on “No-risk” Murchison Island in 1993
762 (Martin et al. 1995) indeed recorded less than 4% of ligneous and herbaceous vegetation cover
763 in the 0 to 1.5 m layer, making it one of the least vegetated understories in the archipelago [see
764 Fig. 2 in Martin et al. (1995), an observation confirmed by more recent visits to Murchison,
765 also see Chollet et al. (2015)]. In contrast to the other islands, Murchison and Faraday islands
766 had been subjected to extensive logging in the first half of the 20th century, resulting in a dense
767 canopy, further reducing forage for deer. Such a high proportion of seaweed in ruminant diet
768 is not unheard of, in particular in conditions where other resources were scarce or unavailable.
769 Seaweeds accounted for up to 50, and even 100%, of the diet of some sheep in primitive North
770 Ronaldsay (Orkey islands) sheep (Balasse et al. 2005) that were confined to the shoreline.

771 However, the two situations with the lowest and similar proportion of seaweed isotopes in
772 jawbone collagen, “Hunting Graham” on Haida Gwaii, and “Predators + hunting” on the
773 coastal mainland, show a dramatic contrast in their cover of understory vegetation. On Yeo
774 Island the permanent presence of predators and hunting is associated with the presence of a
775 lush understory. On Graham the long history of moderate hunting, restricted to areas most
776 accessible to a limited number of hunters (Martin and Baltzinger 2002), had a limited impact
777 on deer density (Engelstoft 2001, Engelstoft et al. 2008) and Graham understories were closer
778 to those on islands with deer but no hunting, than to Yeo understories where predators and
779 hunting occur (Figs. 2 and 8) (Pojar 1999, Chollet et al. 2021b).

780 Post-cull Reef Island represented an intermediate situation. The severe episode of hunting
781 during the experimental culls resulted in a severe, (temporary) reduction in deer abundance
782 triggering a rebound of understory vegetation; an upsurge of resources that could arguably
783 explain the decrease observed in the proportion of seaweeds in their diet after the culls.

784 However, given the lack of a **perfect correlation** between the proportion of seaweeds in deer
785 diet and the abundance of forage in the understory, our results are still consistent with a
786 prediction of an increased reluctance to use more exposed habitats or localities when risk is
787 higher (Williams et al. 2008, Bonnot et al. 2017). The regular decrease in seaweed signature in
788 bone collagen when going from islands without hunting history to islands where current
789 hunting and predators occur, and the reduced seaweed signature in the bones of deer from
790 Graham (hunting) and coastal mainland (predators and hunting), despite their contrast in the
791 abundance of understory vegetation, are suggestive that the risk history on a site negatively
792 affects deer propensity to feed on shorelines a habitat repeatedly shown of high value and use
793 for foraging wolves (Klein 1995, Darimont and Reimchen 2002, Roffler et al. 2023).

794 **Our results from the stable isotope analysis also documented (Fig. 6) a preference for deciduous**
795 **vegetation in deer diet (Taylor 1956, Pojar 1999) that exceeded its relative abundance in the**
796 **understory (Fig. 3).**

797 **What lessons for ecology, population behavioural change and conservation?**

798 Our research adds to the small number of studies that attempted to integrate animal behaviour
799 with the study of the mechanisms that govern ecological dynamics, species interactions,
800 population change and ecosystems [see (Sih et al. 2004, 2012, Réale et al. 2007, Chitwood et
801 al. 2022)]. **We demonstrate that the introduction of a novel threat (i.e., hunting on Reef Island)**
802 **caused an abrupt change in behaviour which afterwards left the norm for the survivors far**
803 **outside the range of observed behaviour on islands without hunting – the sort of behaviour seen**
804 **for Reef Island deer post-cull was never witnessed for deer never exposed to risk on Kunga or**
805 **East Limestone islands. The survivors on Reef probably included animals that modified their**
806 **behaviour more rapidly than those that were culled and, if some or all of that variation in**
807 behaviour was under genetic control (Réale et al. 2007), subsequent generations will have been
808 shyer than the one that was culled. The consequent increases in FID, Dtravel, novelty aversion
809 and nocturnal foraging have persisted and been inherited or adopted by young reared
810 subsequently, so that the behaviour of the current population resembles that of populations
811 subject to natural predation and human hunting. However, there is little sign that the deer
812 translocated from Kunga Island and hence never exposed to predation threat, adopted any of
813 the modified behaviour of the local post-cull deer. This interpretation of a rapid change in
814 dominant behaviours in a population echoes the recent documentation of rapid differential
815 selection of inheritable antipredator behaviours and physical traits in a mammal in response to
816 novel predation (Moseby et al. 2023)

817 Our results are consistent with the hypothesis that changes in the landscape of fear (Gaynor et
818 al. 2019, Palmer et al. 2022) can alter the behaviour characteristic of a population. Such
819 changes can result from severe differential selection of certain behavioural syndromes driven
820 by acute risk (e.g. as in the case of severe culling), or, over longer periods of time, from the

821 progressive reproductive advantage of behavioural profiles better adapted to a changing
822 environment (e.g. driven by a progressive depletion of resources in absence of acute predation
823 risk). Although this remains speculative in the case of the non-wary behaviours we observed
824 on islands without predation or hunting, it illustrates a potential for long term selection. The
825 rapidity of adaptive behavioural change will depend on the magnitude of the selective pressure
826 (Blumstein 2002, Jolly et al. 2018, Moseby et al. 2023). It can be rapid if severe culling is
827 maintained over generations continuously but will presumably be slower in the case of the
828 relaxation of antipredator behaviours after the colonisation of a predator free environment
829 (Blumstein 2002). The evolution of predator naivety in island organisms provides numerous
830 text-book examples of long-term progressive (or fast) evolutionary shifts in behavioural traits
831 (Darwin 1840, Lack 1968, Blumstein 2002), but see Blumstein and Daniel (2005).

832 Our results on shifts in behaviour within a population when the risk context changes have
833 implications for conservation. They can help mitigate the problems posed by the adjustment of
834 focal species towards a novel threat, or facilitate success in species restoration programs where
835 the release of naïve **captive bred** individuals have to adjust to predators (Moseby et al. 2015,
836 2016, 2023). They can also bring insights to management of overabundant populations through
837 hunting. In such instances, a better grasp of behavioural responses to management actions can
838 help design strategies that prevent species adjusting their spatial distribution and diel rhythms
839 to the threat posed by novel management measures (Williams et al. 2008).

840 Improving our understanding of mechanisms that could, over time, lead to evolutionary shifts,
841 in addition to the multifaceted consequences of loosing species from ecological networks,
842 could help better foreseeing the full impact of neglecting the integrity of species assemblages.
843 It would fuel the arguments necessary to prevent such losses, and encourage the conservation
844 and restoration of fully functioning ecosystems. The role left to modern humans in these
845 systems, in place of the significant influence they certainly had on their dynamics for millennia
846 as gatherers and hunters, remains to be defined.

847 **Ethical note**

848 All research was conducted under the appropriate animal care permits from the Wildlife Act
849 of the Ministry of Natural Resources Operation of British Columbia (No. NA11-68421,
850 approved by Parks Canada Animal Care Task Force research permit No. 9059), and under the
851 Archipelago Management Board of Gwaii Haanas National Park Reserve and Haida Heritage
852 Site research permit No. GWA-2011-8373. All except 3 deer captures were done using box
853 traps specially designed for this project (the exceptions were one individual captured by a
854 clover trap and two by a netgun). We made over 400 deer captures or recaptures in the course
855 of the project. Traps were radio-monitored remotely and we dealt with captured animal
856 immediately after capture notification. The animals captured at night were treated the following
857 morning. We recorded no injury to deer, nor mortality associated to capture. The project

858 permits included the experimental culls on one island and the translocation of a small number
859 of deer from one island to another. Culls were conducted by local hunters under clear ethical
860 and practical rules. Carcasses were processed and the meat distributed to the local
861 communities.

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878 **Data, scripts, code, and supplementary information availability**

879 <https://zenodo.org/me/uploads?q=&l=list&p=1&s=10&sort=newest>

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886 **Conflict of interest disclosure**

887 The authors declare that they comply with the PCI rule of having no financial conflicts of
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890

891 **References**

- 892 Altendorf, K. B., Laundré, J. W., López González, C. A., & Brown, J. S. (2001). Assessing effects of
893 predation risk on foraging behavior of mule deer. *Journal of Mammalogy*, 82(2), 430–439.
- 894 Balasse, M., Tresset, A., Dobney, K., & Ambrose, S. H. (2005). The use of isotope ratios to test for seaweed
895 eating in sheep. *Journal of Zoology*, 266(3), 283–291.
- 896 Ballard, W. B., Lutz, D., Keegan, T. W., Carpenter, L. H., & deVos Jr, J. C. (2001). Deer-predator
897 relationships: A review of recent North American studies with emphasis on mule and black-tailed deer.
898 *Wildlife Society Bulletin*, 99–115.
- 899 Banner, A., MacKenzie, W. H., Pojar, J., MacKinnon, A., Saunders, S. C., & Klassen, H. (2014). A field
900 guide to ecosystem classification and identification for Haida Gwaii. *Land Management Handbook*.
- 901 Blumstein, D. T. (2002). Moving to suburbia: Ontogenetic and evolutionary consequences of life on
902 predator-free islands. *Journal of Biogeography*, 29(5-6), 685–692.
- 903 Blumstein, D. T., & Daniel, J. C. (2005). The loss of anti-predator behaviour following isolation on islands.
904 *Proceedings of the Royal Society B: Biological Sciences*, 272(1573), 1663–1668.
- 905 Bocherens, H. (1999). Isotopes stables et reconstitution du régime alimentaire des hominidés fossiles: Une
906 revue. *Bulletins et Mémoires de La Société d'Anthropologie de Paris*, 11(3), 261–287.
- 907 Bocherens, H., & Drucker, D. (2003). Trophic level isotopic enrichment of carbon and nitrogen in bone
908 collagen: Case studies from recent and ancient terrestrial ecosystems. *International Journal of*
909 *Osteoarchaeology*, 13(1-2), 46–53.
- 910 Bocherens, H., Fizet, M., Cuif, J.-P., Jaeger, J.-J., & Michard, J.-G. (1988). Premières mesures
911 d'abondance isotopiques naturelles en ^{13}C et ^{15}N de la matière organique fossile de Dinosaur.
912 Application à l'étude du régime alimentaire du genre *Anatosaurus* (Ornithischia, Hadrosauridae).
913 *Comptes Rendus de l'Académie Des Sciences. Série 2, Mécanique, Physique, Chimie, Sciences de*
914 *l'univers, Sciences de La Terre*, 306(20), 1521–1525.
- 915 Bochéréns, Hervé, Marc Fizet, Lange-Badre, B., Vandermeersch, B., Borel, J. P., & Bellon, G. (1991).
916 Isotopic biogeochemistry (^{13}C , ^{15}N) of fossil vertebrate collagen: Application to the study of a past
917 food web including Neandertal man. *Journal of Human Evolution*, 20(6), 481–492.
- 918 Bøhn, T., & Amundsen, P.-A. (2004). Ecological interactions and evolution: Forgotten parts of
919 biodiversity? *BioScience*, 54(9), 804–805.
- 920 Bonnot, N. C., Couriot, O., Berger, A., Cagnacci, F., Ciuti, S., De Groeve, J. E., Gehr, B., Heurich, M.,
921 Kjellander, P., & Kröschel, M. (2020). Fear of the dark? Contrasting impacts of humans versus lynx on
922 diel activity of roe deer across Europe. *Journal of Animal Ecology*, 89(1), 132–145.
- 923 Bonnot, N. C., Hewison, A. M., Morellet, N., Gaillard, J.-M., Debeffe, L., Couriot, O., Cargnelutti, B.,
924 Chaval, Y., Lourtet, B., & Kjellander, P. (2017). Stick or twist: Roe deer adjust their flight behaviour
925 to the perceived trade-off between risk and reward. *Animal Behaviour*, 124, 35–46.
- 926 Bonnot, N. C., Morellet, N., Hewison, A. M., Martin, J.-L., Benhamou, S., & Chamaillé-Jammes, S.
927 (2016). Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) adjust habitat selection and activity
928 rhythm to the absence of predators. *Canadian Journal of Zoology*, 94(6), 385–394.
- 929 Brown, J. S., Laundré, J. W., & Gurung, M. (1999). The ecology of fear: Optimal foraging, game theory,
930 and trophic interactions. *Journal of Mammalogy*, 80(2), 385–399.
- 931 Burgess, B. T., Irvine, R. L., Martin, J.-L., & Russello, M. A. (2023). Past population control biases
932 interpretations of contemporary genetic data: Implications for future invasive Sitka black-tailed deer
933 management in Haida Gwaii. *Biological Invasions*, 25(12), 3871–3886.
- 934 Burgess, B. T., Irvine, R. L., & Russello, M. A. (2022a). A genotyping-in-thousands by sequencing panel
935 to inform invasive deer management using noninvasive fecal and hair samples. *Ecology and Evolution*,
936 12(6), e8993.
- 937 Burgess, B. T., Irvine, R. L., & Russello, M. A. (2022b). Population genomics of Sitka black-tailed deer
938 supports invasive species management and ecological restoration on islands. *Communications Biology*,
939 5(1), 223.
- 940 Callan, R., Nibbelink, N. P., Rooney, T. P., Wiedenhoeft, J. E., & Wydeven, A. P. (2013). Recolonizing
941 wolves trigger a trophic cascade in Wisconsin (USA). *Journal of Ecology*, 101(4), 837–845.
- 942 <https://doi.org/10.1111/1365-2745.12095>

943 Camin, F., Bontempo, L., Perini, M., & Piasentier, E. (2016). Stable isotope ratio analysis for assessing
944 the authenticity of food of animal origin. *Comprehensive Reviews in Food Science and Food Safety*,
945 15(5), 868–877.

946 Cardinal, E., Martin, J.-L., & Côté, S. D. (2012). Large herbivore effects on songbirds in boreal forests:
947 Lessons from deer introduction on Anticosti Island. *Ecoscience*, 19(1), 38–47.

948 Cardinal, E., Martin, J.-L., Tremblay, J.-P., & Côté, S. D. (2012). An experimental study of how variation
949 in deer density affects vegetation and songbird assemblages of recently harvested boreal forests.
950 *Canadian Journal of Zoology*, 90(6), 704–713.

951 Chamaillé-Jammes, S., Malcuit, H., Le Saout, S., & Martin, J.-L. (2014). Innate threat-sensitive foraging:
952 Black-tailed deer remain more fearful of wolf than of the less dangerous black bear even after 100 years
953 of wolf absence. *Oecologia*, 174(4), 1151–1158.

954 Chitwood, M. C., Baruzzi, C., & Lashley, M. A. (2022). “Ecology of fear” in ungulates: Opportunities for
955 improving conservation. *Ecology and Evolution*, 12(3), e8657.

956 Chollet, S. (2012). De la mise en évidence à la gestion de l’effet de cerf Leçons pratiques et théoriques
957 fournies par l’introduction du cerf à queue-noire sur Haïda Gwaii. Université Montpellier II-Sciences
958 et Techniques du Languedoc.

959 Chollet, S., Baltzinger, C., Maillard, M., & Martin, J.-L. (2021b). Deer exclusion unveils abiotic filtering
960 in forest understorey plant assemblages. *Annals of Botany*, 128(3), 371–381.

961 Chollet, S., Baltzinger, C., Ostermann, L., Saint-André, F., & Martin, J.-L. (2013). Importance for forest
962 plant communities of refuges protecting from deer browsing. *Forest Ecology and Management*, 289,
963 470–477. <https://doi.org/10.1016/j.foreco.2012.10.043>

964 Chollet, S., Bergman, C., Gaston, A. J., & Martin, J.-L. (2015). Long-term consequences of invasive deer
965 on songbird communities: Going from bad to worse? *Biological Invasions*, 17(2), 777–790.

966 Chollet, S., Maillard, M., Schörghuber, J., Grayston, S. J., & Martin, J.-L. (2021a). Deer slow down litter
967 decomposition by reducing litter quality in a temperate forest. *Ecology*, 102(2), e03235.

968 Chollet, S., Padié, S., Stockton, S., Allombert, S., Gaston, A. J., & Martin, J.-L. (2016). Positive plant and
969 bird diversity response to experimental deer population reduction after decades of uncontrolled
970 browsing. *Diversity and Distributions*, 22(3), 274–287. <https://doi.org/10.1111/ddi.12393>

971 Cooper Jr, W. E. (2008). Visual monitoring of predators: Occurrence, cost and benefit for escape. *Animal*
972 *Behaviour*, 76(4), 1365–1372.

973 Côté, S. D., Rooney, T. P., Tremblay, J.-P., Dussault, C., & Waller, D. M. (2004). Ecological Impacts of
974 Deer Overabundance. *Annual Review of Ecology, Evolution, and Systematics*, 35, 113–147.

975 Darimont, C. T., Paquet, P. C., & Reimchen, T. E. (2007). Stable isotopic niche predicts fitness of prey in
976 a wolf–deer system. *Biological Journal of the Linnean Society*, 90(1), 125–137.

977 Darimont, C. T., & Reimchen, T. E. (2002). Intra-hair stable isotope analysis implies seasonal shift to
978 salmon in gray wolf diet. *Canadian Journal of Zoology*, 80(9), 1638–1642.

979 Darwin, C. (1840). *Journal of Researches into the Geology and Natural History of the Various Countries*
980 *Visited by HMS Beagle, under the Command of Captain Fitzroy from 1832 to 1836 by Charles Darwin.*
981 Colburn.

982 Daufresne, T., & Martin, J. L. (1997). Changes in vegetation structure and diversity as a result of browsing
983 by a large herbivore: The impact of introduced Black-tailed deer in the primary forest of Haida Gwaii,
984 British Columbia. *Laskeek Bay Research*, 7, 2–26.

985 DeNiro, M. J., & Epstein, S. (1978). Influence of diet on the distribution of carbon isotopes in animals.
986 *Geochimica et Cosmochimica Acta*, 42(5), 495–506.

987 DeNiro, M. J., & Epstein, S. (1981). Influence of diet on the distribution of nitrogen isotopes in animals.
988 *Geochimica et Cosmochimica Acta*, 45(3), 341–351.

989 Eilers, J., Toby Kiers, E., Currie, C. R., McDonald, B. R., & Visser, B. (2012). Ecological interactions
990 drive evolutionary loss of traits. *Ecology Letters*, 15(10), 1071–1082.

991 Elton, C. (1927). *Animal ecology*. 207 pp. Sidgwick & Jackson, LTD. London.

992 Engelstoft, C. (2001). Effects of Sitka Black-tailed Deer (*Odocoileus Hemionus Sitkensis*) on Understorey
993 in Old Growth Forest on Haida Gwaii (Queen Charlotte Islands), British Columbia. University of
994 Victoria.

995 Engelstoft, C., Kirchhoff, M. D., & Eastman, D. (2008). Herbivory and the missing understorey on Haida
996 Gwaii. In *Lessons from the Islands: Introduced Species and What They Tell Us About How Ecosystems*
997 *Work*. (pp. 78–86). Canadian Wildlife Service, Environment Canada, Ottawa.

998 Estes, J. A., Brashares, J. S., & Power, M. E. (2013). Predicting and detecting reciprocity between indirect
999 ecological interactions and evolution. *The American Naturalist*, 181(S1), S76–S99.

1000 Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., Carpenter, S. R.,
1001 Essington, T. E., Holt, R. D., Jackson, J. B. C., Marquis, R. J., Oksanen, L., Oksanen, T., Paine, R. T.,
1002 Pickett, E. K., Ripple, W. J., Sandin, S. A., Scheffer, M., Schoener, T. W., ... Wardle, D. A. (2011).
1003 Trophic Downgrading of Planet Earth. *Science*, 333(6040), 301–306.

1004 Gaston, A. J., Stockton, S. A., & Smith, J. L. (2006). Species-area relationships and the impact of deer-
1005 browse in the complex phytogeography of the Haida Gwaii archipelago (Queen Charlotte Islands),
1006 British Columbia. *Écoscience*, 13(4), 511–522. [https://doi.org/10.2980/1195-
1007 6860\(2006\)13\[511:SRATIO\]2.0.CO;2](https://doi.org/10.2980/1195-6860(2006)13[511:SRATIO]2.0.CO;2)

1008 Gaynor, K. M., Brown, J. S., Middleton, A. D., Power, M. E., & Brashares, J. S. (2019). Landscapes of
1009 fear: Spatial patterns of risk perception and response. *Trends in Ecology & Evolution*, 34(4), 355–368.

1010 Golumba, T., Bland, L., Moore, K., & Bartier, P. (2008). History and current status of introduced
1011 vertebrates on Haida Gwaii. In *Lessons from the islands: Introduced species and what they tell us about
1012 how ecosystems work* (pp. 8–31). Canadian Wildlife Service, Environment Canada, Ottawa.

1013 Greenberg, R., & Mettke-Hofmann, C. (2001). Ecological aspects of neophobia and neophilia in birds.
1014 *Current Ornithology*, 119–178.

1015 Haeckel, E. (1866). *Generelle Morphologie der Organismen. Allgemeine Grundzüge der organischen
1016 Formen-Wissenschaft, mechanisch begründet durch die von C. Darwin reformirte Descendenz-Theorie.
1017 Allgemeine Entwicklungsgeschichte.* (Vol. 2). De Gruyter.

1018 Janzen, D. H. (1974). The deflowering of central america. *Natural History Magazine*, 83(4), 48–51.

1019 Jolly, C. J., Webb, J. K., & Phillips, B. L. (2018). The perils of paradise: An endangered species conserved
1020 on an island loses antipredator behaviours within 13 generations. *Biology Letters*, 14(6), 20180222.
1021 <https://doi.org/10.1098/rsbl.2018.0222>

1022 Kelly, J. F. (2000). Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic
1023 ecology. *Canadian Journal of Zoology*, 78(1), 1–27.

1024 Klein, D. R. (1995). The introduction, increase, and demise of wolves on Coronation Island, Alaska.
1025 *Ecology and Conservation of Wolves in a Changing World*, 275, 280.

1026 Kuijper, D. P. J., Sahlén, E., Elmhagen, B., Chamaillé-Jammes, S., Sand, H., Lone, K., & Cromsigt, J.
1027 (2016). Paws without claws? Ecological effects of large carnivores in anthropogenic landscapes. *Proc.
1028 R. Soc. B*, 283. <http://dx.doi.org/10.1098/rspb.2016.162>

1029 Lack, D. L. (1968). *Ecological adaptations for breeding in birds.* Methuen.

1030 Le Saout, S., Chollet, S., Chamaillé-Jammes, S., Blanc, L., Padié, S., Verchere, T., Gaston, A. J.,
1031 Gillingham, M. P., Gimenez, O., Parker, K. L., Picot, Verheyden, & Martin, J.-L. (2014). Understanding
1032 the paradox of deer persisting at high abundance in heavily browsed habitats. *Wildlife Biology*, 20(3),
1033 122–135.

1034 Le Saout, S., Martin, J.-L., Blanchard, P., Cebe, N., Mark Hewison, A. J., Rames, J.-L., & Chamaillé-
1035 Jammes, S. (2015). Seeing a ghost? Vigilance and its drivers in a predator-free world. *Ethology*, 121(7),
1036 651–660.

1037 Lima, S. L., & Dill, L. M. (1990). Behavioral decisions made under the risk of predation: A review and
1038 prospectus. *Canadian Journal of Zoology*, 68(4), 619–640.

1039 Maillard, M., Martin, J.-L., Chollet, S., Catomeris, C., Simon, L., & Grayston, S. J. (2021). Belowground
1040 effects of deer in a temperate forest are time-dependent. *Forest Ecology and Management*, 493, 119228.

1041 Martin, J.-L., & Baltzinger, C. (2002). Interaction among deer browsing, hunting, and tree regeneration.
1042 *Canadian Journal of Forest Research*, 32(7), 1254–1264. <https://doi.org/10.1139/x02-043>

1043 Martin, J.-L., Chamaillé-Jammes, S., & Waller, D. M. (2020). Deer, wolves, and people: Costs, benefits
1044 and challenges of living together. *Biological Reviews*, 93(3), 782–801.

1045 Martin, J.-L., Gaston, A. J., & Hitier, S. (1995). The effect of island size and isolation on old growth forest
1046 habitat and bird diversity in Gwaii Haanas (Queen Charlotte Islands, Canada). *Oikos*, 115–131.

1047 Martin, J.-L., Stockton, S. A., Allombert, S., & Gaston, A. J. (2010). Top-down and bottom-up
1048 consequences of unchecked ungulate browsing on plant and animal diversity in temperate forests:
1049 Lessons from a deer introduction. *Biological Invasions*, 12(2), 353–371.
1050 <https://doi.org/10.1007/s10530-009-9628-8>

- 1051 Martin, T. G., Arcese, P., & Scheerder, N. (2011). Browsing down our natural heritage: Deer impacts on
1052 vegetation structure and songbird populations across an island archipelago. *Biological Conservation*,
1053 144(1), 459–469.
- 1054 Mathews, N. E., & Porter, W. F. (1988). Black bear predation of white-tailed deer neonates in the central
1055 Adirondacks. *Canadian Journal of Zoology*, 66(5), 1241–1242.
- 1056 Monestier, C., Morellet, N., Verheyden, H., Gaillard, J.-M., Bideau, E., Denailhac, A., Lourtet, B., Cebe,
1057 N., Picot, D., & Rames, J.-L. (2017). Neophobia is linked to behavioural and haematological indicators
1058 of stress in captive roe deer. *Animal Behaviour*, 126, 135–143.
- 1059 Moseby, K. E., Blumstein, D. T., & Letnic, M. (2016). Harnessing natural selection to tackle the problem
1060 of prey naïveté. *Evolutionary Applications*, 9(2), 334–343.
- 1061 Moseby, K. E., Carthey, A., & Schroeder, T. (2015). The influence of predators and prey naivety on
1062 reintroduction success: Current and future directions. *Advances in Reintroduction Biology of*
1063 *Australian and New Zealand Fauna*, 29–42.
- 1064 Moseby, K., Van der Weyde, L., Letnic, M., Blumstein, D. T., West, R., & Bannister, H. (2023).
1065 Addressing prey naivety in native mammals by accelerating selection for antipredator traits. *Ecological*
1066 *Applications*, 33(2), e2780.
- 1067 Paine, R. T. (1966). Food web complexity and species diversity. *The American Naturalist*, 100(910), 65–
1068 75.
- 1069 Paine, R. T. (1969). A Note on Trophic Complexity and Community Stability. *The American Naturalist*,
1070 103(929), 91–93.
- 1071 Palmer, M. S., Gaynor, K. M., Becker, J. A., Abraham, J. O., Mumma, M. A., & Pringle, R. M. (2022).
1072 Dynamic landscapes of fear: Understanding spatiotemporal risk. *Trends in Ecology & Evolution*,
1073 37(10). <https://doi.org/10.1016/j.tree.2022.06.007>
- 1074 Pojar, J. (1999). The effects of deer browsing on the plant life of Haida Gwaii. *Proceedings of the Cedar*
1075 *Symposium: Growing Western Redcedar and Yellowcypress on the Queen Charlotte Islands/Haida*
1076 *Gwaii*. BC Ministry of Forests/Canada–British Columbia South Moresby Forest Replacement Account,
1077 Victoria, BC, 90–97.
- 1078 Racine, J. S. (2012). RStudio: A platform-independent IDE for R and Sweave. JSTOR.
- 1079 Réale, D., Garant, D., Humphries, M. M., Bergeron, P., Careau, V., & Montiglio, P.-O. (2010). Personality
1080 and the emergence of the pace-of-life syndrome concept at the population level. *Philosophical*
1081 *Transactions of the Royal Society B: Biological Sciences*, 365(1560), 4051–4063.
- 1082 Réale, D., Reader, S. M., Sol, D., McDougall, P. T., & Dingemanse, N. J. (2007). Integrating animal
1083 temperament within ecology and evolution. *Biological Reviews*, 82(2), 291–318.
- 1084 Richards, M. P., Fuller, B. T., & Molleson, T. I. (2006). Stable isotope palaeodietary study of humans and
1085 fauna from the multi-period (Iron Age, Viking and Late Medieval) site of Newark Bay, Orkney. *Journal*
1086 *of Archaeological Science*, 33(1), 122–131.
- 1087 Rodriguez-Prieto, I., Fernández-Juricic, E., Martín, J., & Regis, Y. (2009). Antipredator behavior in
1088 blackbirds: Habituation complements risk allocation. *Behavioral Ecology*, 20(2), 371–377.
- 1089 Roffler, G. H., Eriksson, C. E., Allen, J. M., & Levi, T. (2023). Recovery of a marine keystone predator
1090 transforms terrestrial predator–prey dynamics. *Proceedings of the National Academy of Sciences*,
1091 120(5), e2209037120.
- 1092 Salomon, A. K., Shears, N. T., Langlois, T. J., & Babcock, R. C. (2008). Cascading effects of fishing can
1093 alter carbon flow through a temperate coastal ecosystem. *Ecological Applications*, 18(8), 1874–1887.
- 1094 Schulting, R. J., Blockley, S. M., Bocherens, H., Drucker, D., & Richards, M. (2008). Stable carbon and
1095 nitrogen isotope analysis on human remains from the Early Mesolithic site of La Vergne (Charente-
1096 Maritime, France). *Journal of Archaeological Science*, 35(3), 763–772.
- 1097 Schuttler, S. G., Parsons, A. W., Forrester, T. D., Baker, M. C., McShea, W. J., Costello, R., & Kays, R.
1098 (2017). Deer on the lookout: How hunting, hiking and coyotes affect white-tailed deer vigilance. *Journal*
1099 *of Zoology*, 301(4), 320–327.
- 1100 Sih, A., Bell, A., & Johnson, J. C. (2004). Behavioral syndromes: An ecological and evolutionary
1101 overview. *Trends in Ecology & Evolution*, 19(7), 372–378.
- 1102 Sih, A., Cote, J., Evans, M., Fogarty, S., & Pruitt, J. (2012). Ecological implications of behavioural
1103 syndromes. *Ecology Letters*, 15(3), 278–289.
- 1104 Soulé, M. E., Estes, J. A., Berger, J., & Del Rio, C. M. (2003). Ecological effectiveness: Conservation
1105 goals for interactive species. *Conservation Biology*, 17(5), 1238–1250.

1106 Stankowich, T. (2008). Ungulate flight responses to human disturbance: A review and meta-analysis.
1107 *Biological Conservation*, 141(9), 2159–2173.

1108 Stankowich, T., & Coss, R. G. (2006). Effects of predator behavior and proximity on risk assessment by
1109 Columbian black-tailed deer. *Behavioral Ecology*, 17(2), 246–254.

1110 Stankowich, T., & Coss, R. G. (2007). Effects of risk assessment, predator behavior, and habitat on escape
1111 behavior in Columbian black-tailed deer. *Behavioral Ecology*, 18(2), 358–367.

1112 Stock, B. C., Jackson, A. L., Ward, E. J., Parnell, A. C., Phillips, D. L., & Semmens, B. X. (2018).
1113 Analyzing mixing systems using a new generation of Bayesian tracer mixing models. *PeerJ*, 6, e5096.

1114 Stock, B., Semmens, B., Ward, E., Parnell, A., Jackson, A., & Phillips, D. (2018). Package ‘MixSIAR.’
1115 *Bayesian Mixing Models in R*, Version, 3(10).

1116 Stockton, S. A., Allombert, S., Gaston, A. J., & Martin, J.-L. (2005). A natural experiment on the effects
1117 of high deer densities on the native flora of coastal temperate rain forests. *Biological Conservation*,
1118 126(1), 118–128. <https://doi.org/10.1016/j.biocon.2005.06.006>

1119 Taylor, W. P. (1956). *The Deer of North America*. Stackpole Company and Wildlife Management Institute,
1120 Washington.

1121 Team, R. C. (2018). *R: A language and environment for statistical computing*; 2018.

1122 Valiente-Banuet, A., Aizen, M. A., Alcántara, J. M., Arroyo, J., Cocucci, A., Galetti, M., García, M. B.,
1123 García, D., Gómez, J. M., & Jordano, P. (2015). Beyond species loss: The extinction of ecological
1124 interactions in a changing world. *Functional Ecology*, 29(3), 299–307.

1125 Vila, B., Torre, F., Guibal, F., & Martin, J.-L. (2004). Can we reconstruct browsing history and how far
1126 back? Lessons from *Vaccinium parvifolium* Smith in Rees. *Forest Ecology and Management*, 201(2),
1127 171–185.

1128 Waller, D. M., & Reo, N. J. (2018). First stewards: Ecological outcomes of forest and wildlife stewardship
1129 by indigenous peoples of Wisconsin, USA. *Ecology and Society*, 23(1), 45. [https://doi.org/10.5751/ES-](https://doi.org/10.5751/ES-09865-230145)
1130 [09865-230145](https://doi.org/10.5751/ES-09865-230145)

1131 Williams, S. C., DeNicola, A. J., & Ortega, I. M. (2008). Behavioral responses of white-tailed deer
1132 subjected to lethal management. *Canadian Journal of Zoology*, 86(12), 1358–1366.

1133 Ydenberg, R. C., & Dill, L. M. (1986). The economics of fleeing from predators. *Advances in the Study*
1134 *of Behavior*, 16, 229–249.

1135