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

Jean-Louis Martin <sup>\*</sup>1, Simon Chamaillé-Jammes<sup>1</sup>, Anne Salomon<sup>2</sup>, Devana Veronica Gomez Pourroy<sup>1</sup>, Mathilde Schlaeflin<sup>1</sup>, Soizic Le Saout<sup>1</sup>, Annick Lucas<sup>1</sup>, Ilham Bentaleb<sup>3</sup>, Simon Chollet<sup>4</sup>, Jake Pattison<sup>5</sup>, Soline Martin-Blangy<sup>6</sup>, Anthony J. Gaston<sup>7</sup>

<sup>1</sup> CEFE, Univ Montpellier, CNRS, EPHE, IRD - Montpellier, France

<sup>2</sup> School of Resource & Environmental Management, Simon Fraser University - Burnaby, Canada

<sup>3</sup> University of Montpellier, UMR 5554 CNRS/IRD/EPHE, CC061 - Montpellier, France

<sup>4</sup> University of Rennes, CNRS, ECOBIO Ecosystèmes, biodiversité, évolution, UMR 6553 - Rennes, France

<sup>5</sup> Laskeek Bay Conservation Society - Daajing Giids and Skidegate, Canada

<sup>6</sup> 81 rue Marius Carrieu, 34080 Montpellier, France

<sup>7</sup> National Wildlife Research Center, Environment and Climate Change Canada - Ottawa, Canada

\*Jean-Louis Martin

Correspondence: [jean-louis.martin@cefe.cnrs.fr](mailto:jean-louis.martin@cefe.cnrs.fr)

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 / km<sup>2</sup> (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 <sup>2</sup>	Y	N	Y	Y
Risk	Yeo	Coastal BC	95 km <sup>2</sup>	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/km<sup>2</sup>, 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 km<sup>2</sup>), 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/km<sup>2</sup> (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 m<sup>2</sup>) 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 Chamailé-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  $^{13}\text{C}$  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 ( $\delta$ ) 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  $\delta X$  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  $\text{N}_2$ .

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 CO<sub>2</sub> and N<sub>2</sub> 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).

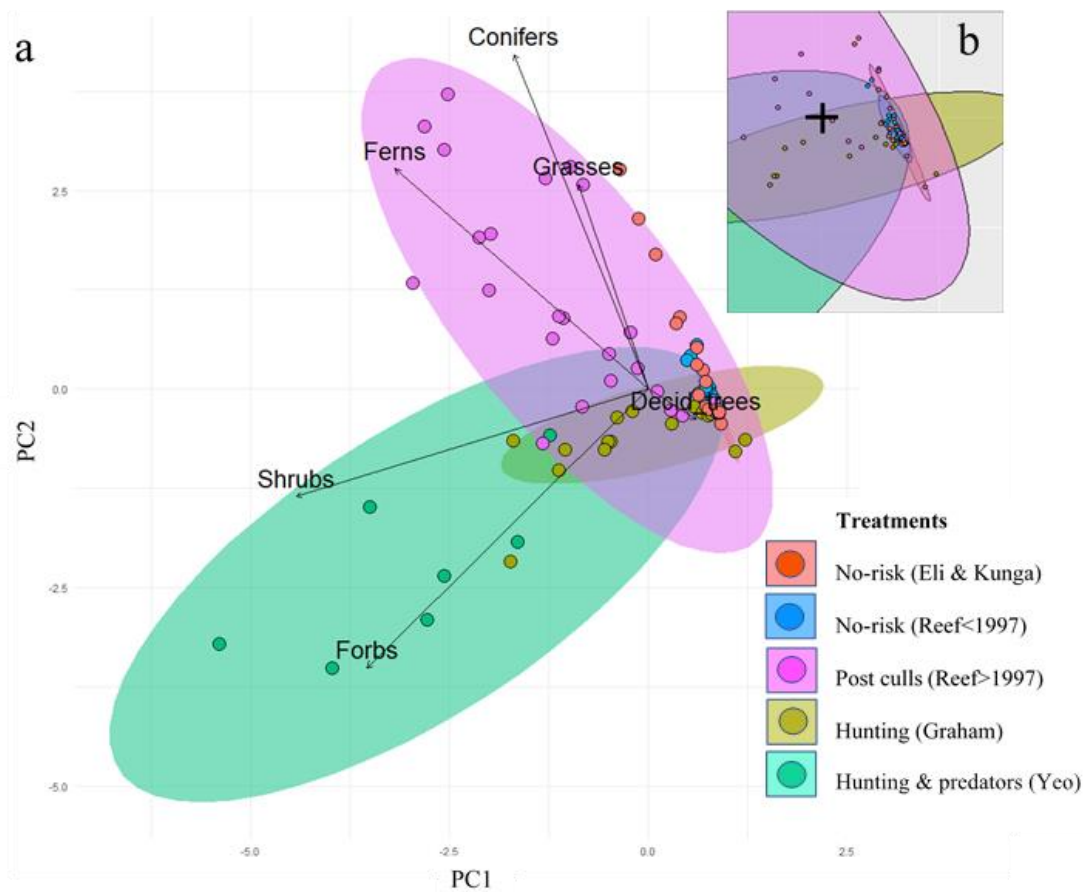
## 350 **Results**

### 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

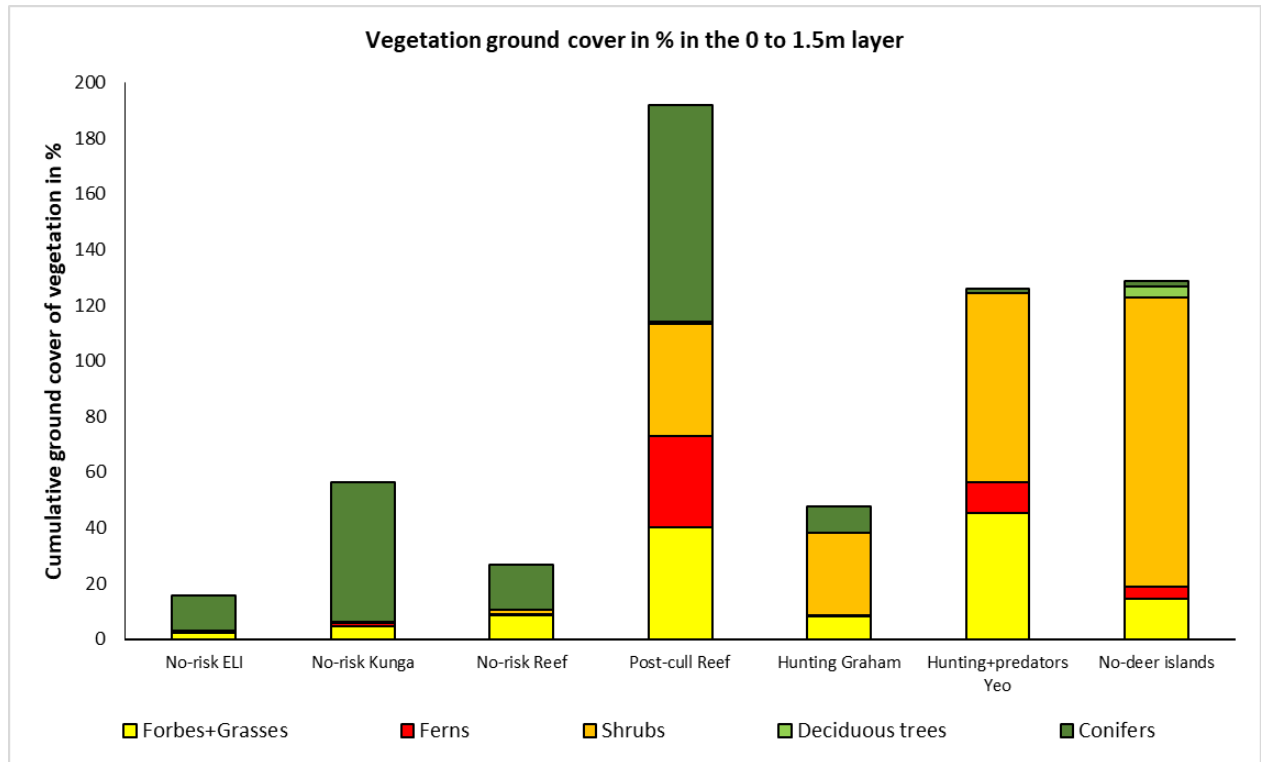


366

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.

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378



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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).

400

401

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

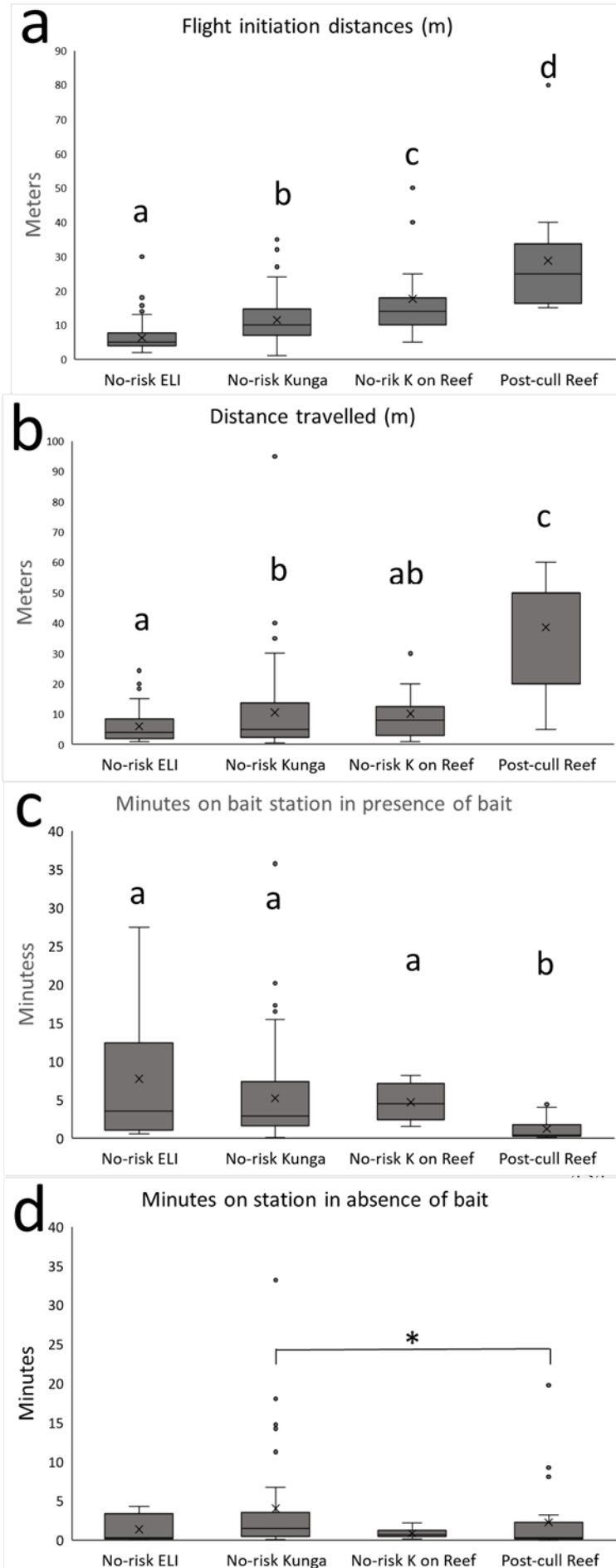
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407 **Table 3.** Flight initiation distances (FID) in meter and distance travelled after flight (Dtravel) in meter before resuming  
 408 activity for the different treatment categories and for marked versus unmarked deer. Events = total number of 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	

413  
 414





**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 (log<sub>10</sub> 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					

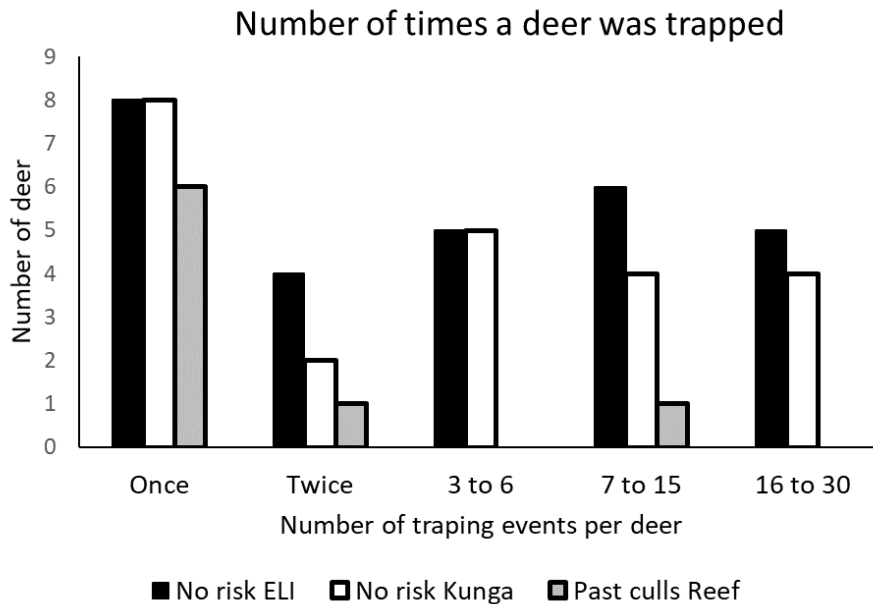
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532 **Response to traps** – On Reef Island post-cull the capture rate was 90% lower than on East  
 533 Limestone and 80% lower than on Kunga (Table 6). Only one third as many animals were captured  
 534 on Reef Island post-cull as on the two “No-risk” 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.

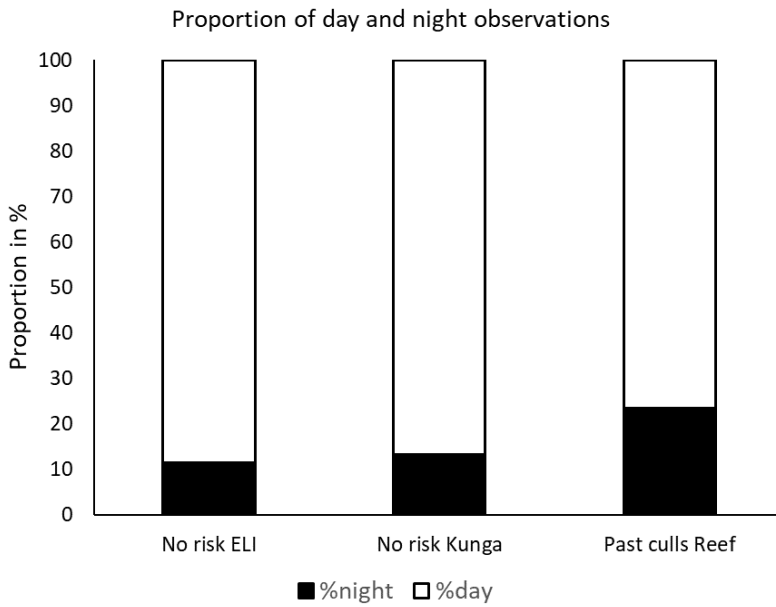
549 **Diel activity pattern varied with risk category**

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)

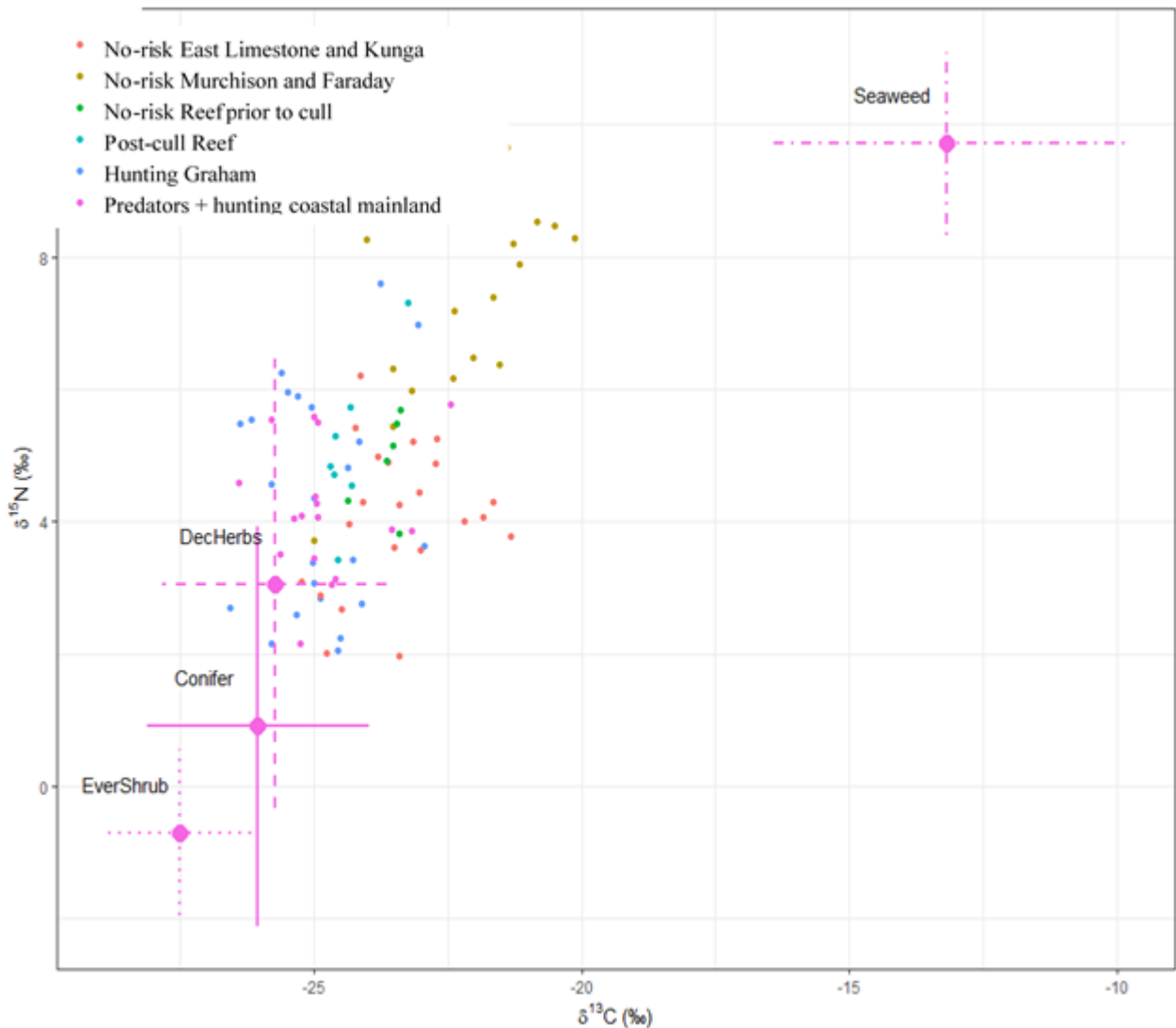
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### 581 **Stable isotopes as indicator of the propensity to use exposed shorelines**

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  $\pm 1.96$  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.

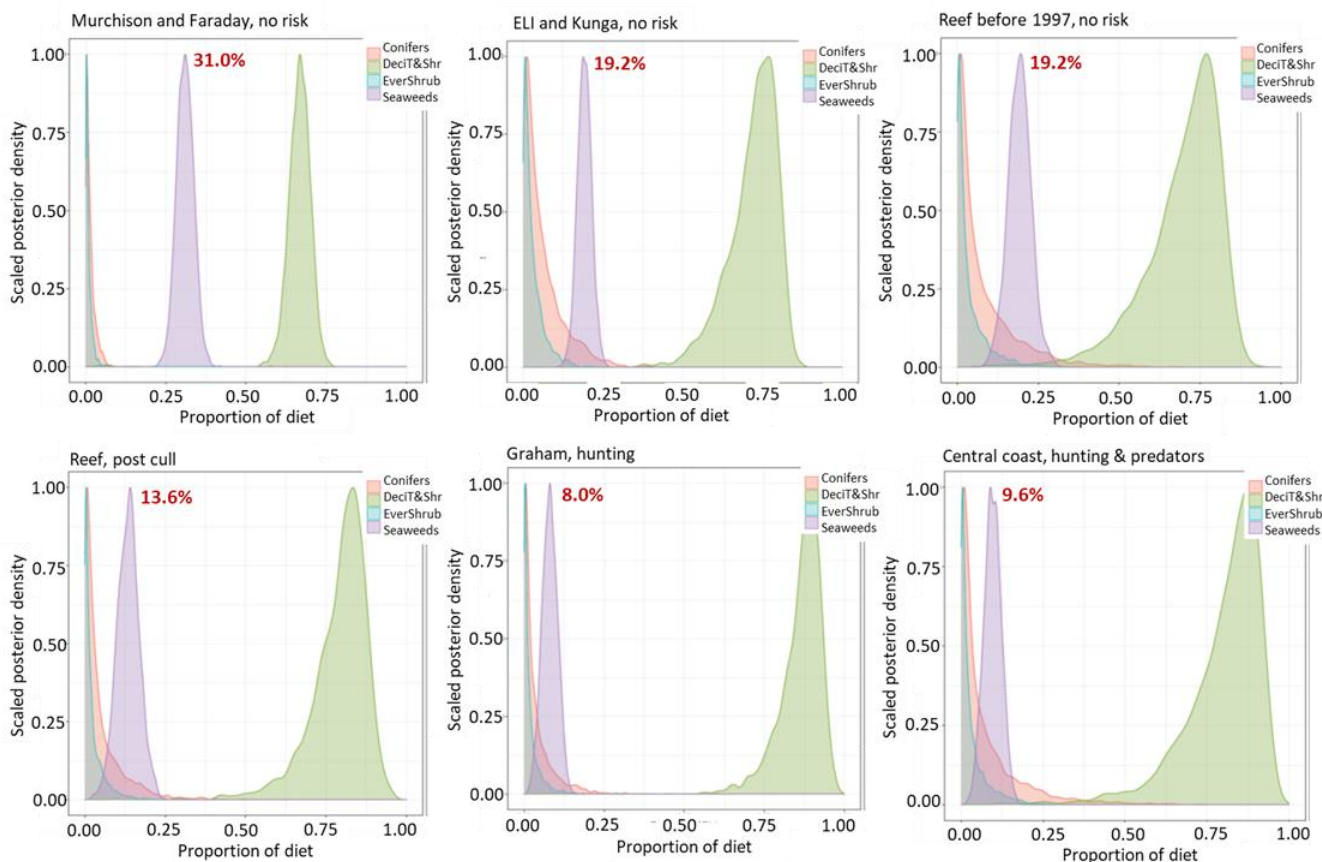


601  
 602 **Figure 7.** Isospace plots of deer and plants Mixture data (deer) by risk category and islands. No-risk Murchison and  
 603 Faraday = No-risk samples from Murchison (N=10 bone samples) and Faraday (N=6); No-risk samples from East  
 604 Limestone (N=10) and Kunga (N=12); No-risk samples 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).

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 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/km<sup>2</sup>, 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 km<sup>2</sup>.  
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  
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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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887 The authors declare that they comply with the PCI rule of having no financial conflicts of  
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890

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