



**HAL**  
open science

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

Jean-Louis Martin, Simon Chamaille-Jammes, Anne Salomon, Devana  
Veronica Gomez Pourroy, Mathilde Schlaeflin, Soizic Le Saout, Annick Lucas,  
Ilham Bentaleb, Simon Chollet, Jake Pattison, et al.

### ► To cite this version:

Jean-Louis Martin, Simon Chamaille-Jammes, Anne Salomon, Devana Veronica Gomez Pourroy, Mathilde Schlaeflin, et al. From fear to food: predation risk shapes deer behaviour, their resources and forest vegetation. 2024. hal-04381108v1

**HAL Id: hal-04381108**

**<https://hal.science/hal-04381108v1>**

Preprint submitted on 9 Jan 2024 (v1), last revised 18 Nov 2024 (v5)

**HAL** is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1  
2  
3  
4  
5  
6  
7  
8  
9  
10  
11  
12  
13  
14  
15  
16  
17  
18  
19  
20  
21  
22  
23

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

Jean-Louis Martin<sup>\*1</sup>, Simon Chamailé-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>, Tony 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> PO Box 792, 408 2nd Ave - Daajing Giids, 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

50

## Introduction

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 19<sup>th</sup> century (Golumbia et al. 2008). They colonized all but a few small islands. The occurrence  
111 of a few reference islands that never supported deer made it possible to demonstrate that, on islands  
112 with long-term deer presence, independent of island size, deer herbivory was the main factor

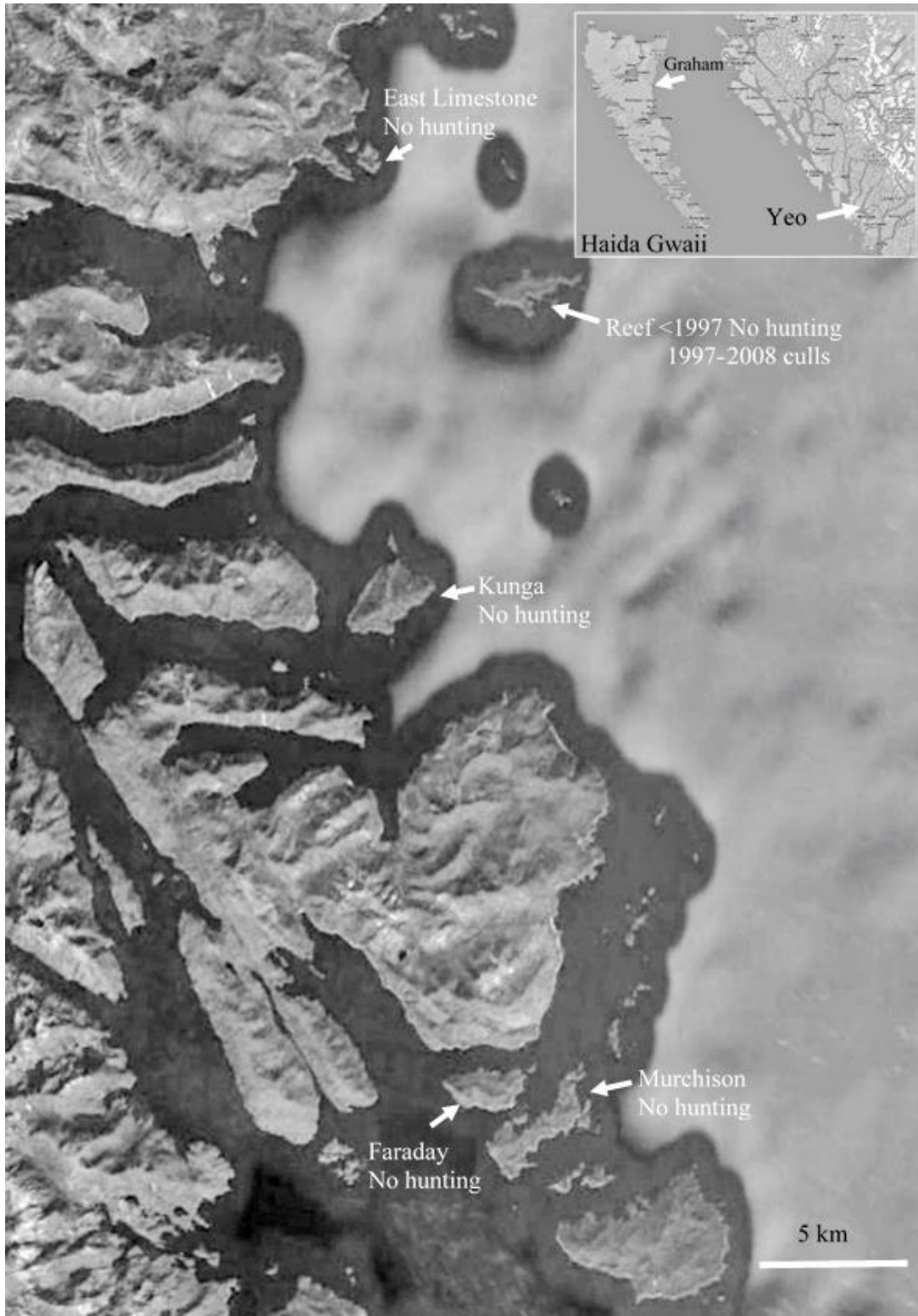
113 structuring plant and animal communities (Martin and Baltzinger 2002, Gaston et al. 2006, Martin  
 114 et al. 2010, Chollet et al. 2013), with important consequences on belowground processes (Chollet  
 115 et al. 2021a, Maillard et al. 2021). Recurrent experimental culls on two islands allowed us to  
 116 monitor the response of the aboveground vegetation and avifauna for 13 years (Chollet et al. 2016).  
 117 The three islands in Laskeek Bay (52°53'12"N, 131°35'20"W) where we focused our study [Reef  
 118 249 ha, Kunga 395 ha, and East Limestone Island 48 ha)] (Table 1) have all had deer present for  
 119 over 60 years at the time of study (Vila et al. 2004) and had no history of hunting (Fig. 1). Their  
 120 initial deer density was estimated at approximately 30 deer / km<sup>2</sup> (Daufresne and Martin 1997).  
 121 Covered by mature forests without human settlements or activities, they were all initially  
 122 characterized by open and species poor understories (Martin et al. 1995, Stockton et al. 2005,  
 123 Martin et al. 2010) representative of severe deer impacts that we documented at the scale of the  
 124 archipelago (Martin et al. 2010, Chollet et al. 2015).

125

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

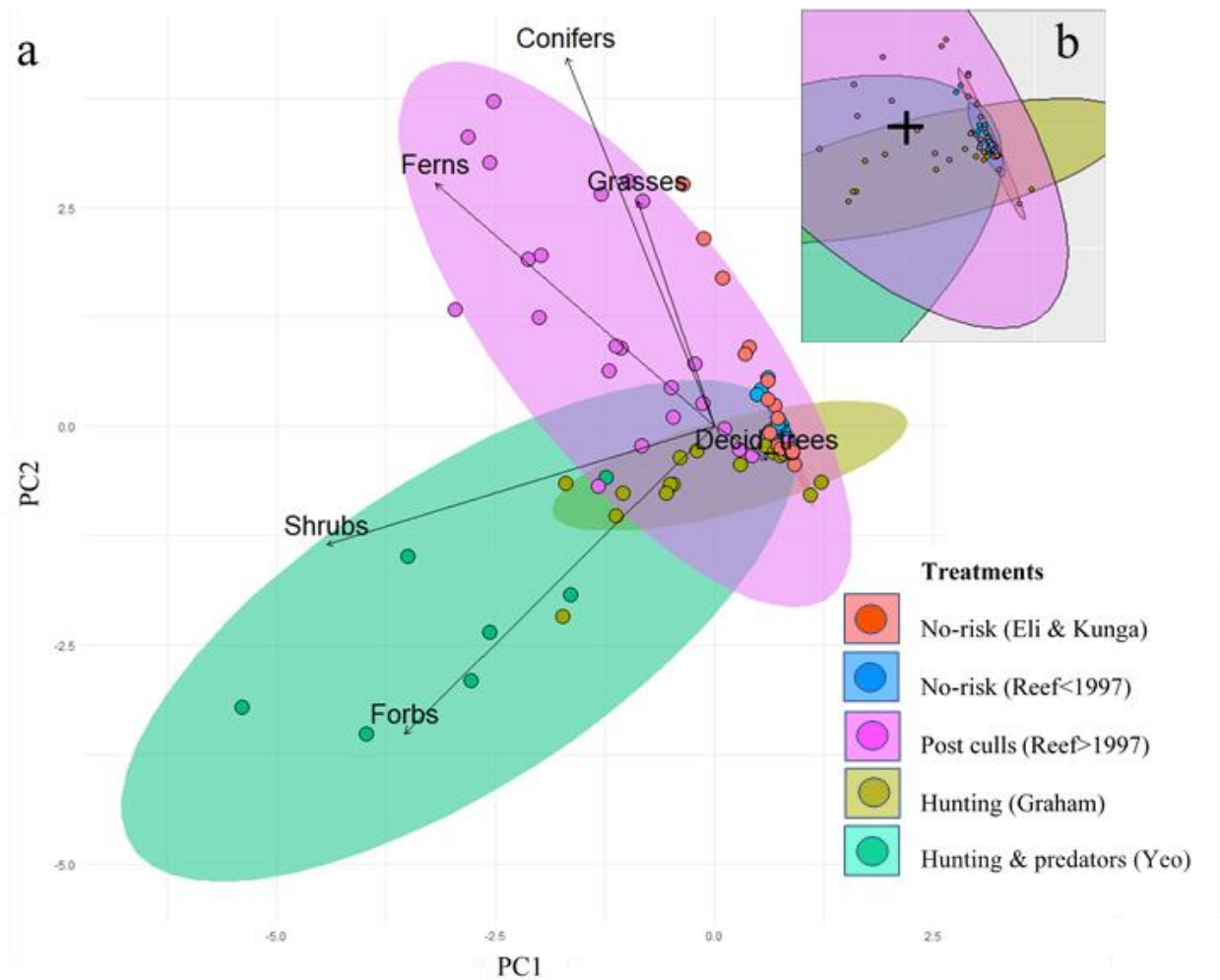
## 350 **Results**

### 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

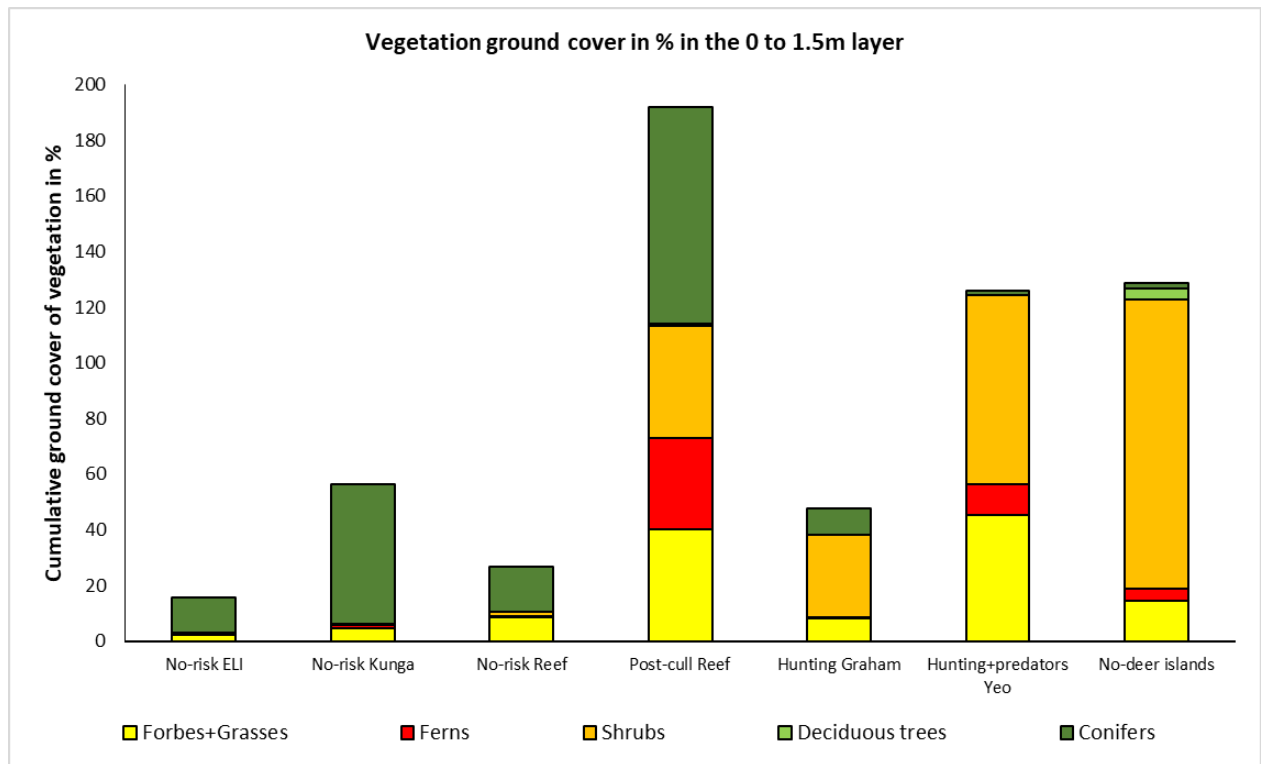
360 Island post-cull sample. Yeo Island plots had a high shrub and forb cover (Figs. 2 and 3), while  
 361 those on Reef Island post-cull showed high cover in conifers, ferns and grasses and an increase in  
 362 forb and shrub cover (Figs. 2 and 3). Among plot variability was lower on Graham Island, 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  
 368 Analysis on understory vegetation in relation to the main categories of understory vegetation. Treatments  
 369 were color coded post analysis. Treatments are: No hunting represented by [No-risk (East Limestone (Eli)  
 370 & Kunga)] = East Limestone and Kunga islands, in red, and [No-risk (Reef<1997)] = Reef island before  
 371 the cull (1997), in blue; Presence of risk through hunting with [Past culls (Reef>1997)] = Reef Island  
 372 after the 1997 to 2008 culls, in pink, and [Hunting (Graham)] = long history of yearly hunting, in kaki;  
 373 Presence of risk through regular hunting and presence of deer natural predators [Hunting + predators  
 374 (Yeo)] = Yeo island, in green. **b:** zoom around plot center showing the ellipses corresponding to the [No-  
 375 risk (Eli & Kunga)] (small red shade) and [No-risk (Reef<1997)] (tiny blue shade). Their small extent  
 376 expresses the extreme similarity among vegetation plots in these samples. Note the five outliers from  
 377 Kunga in the [No-risk (Eli & Kunga)] sample.



380

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

388

389 **Flight initiation distance varied with risk category**

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



398 **Table 2.** Flight initiation distance (FID) in meter: FID\_mean = mean values, FID\_sd = standard  
 399 deviation, and N = sample size for the different deer categories studied.

400

| 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  |

401

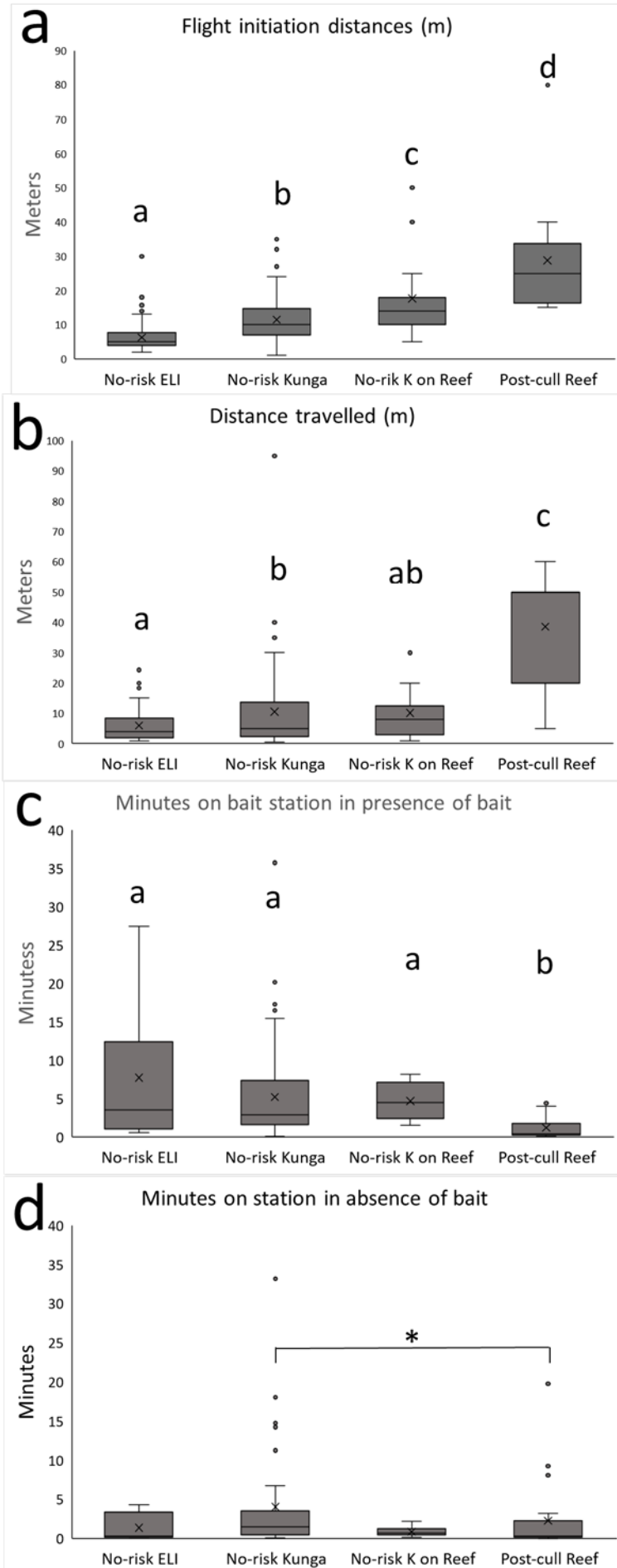
402

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

409

410



**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).

455 **Table 4.** Distance Travelled (Dtravel) after flight initiation: Dtravel\_mean = mean values, Dtravel\_sd =  
 456 standard 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  |

457  
 458 Only the samples from East Limestone and Reef Island (post-cull) included marked deer. There  
 459 was a marginal trend for longer FID in “No-risk” East Limestone marked deer (t-ratio = 1.85, p =  
 460 0.07) (6.9 versus 5.5 m). There was no statistically significant difference in FID between marked  
 461 and unmarked deer in “post-cull Reef” (t-ratio = 0.95, p = 0.34) (36 versus 23.6 m) (Table 3).

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

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

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

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

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

480 When bait was present, deer on islands with no risk, and deer from Kunga translocated to Reef,  
481 spent significantly more time at bait stations than the native Reef Island deer post-cull (Fig. 4c).  
482 The rate of visits at bait stations on islands with no risk history (East Limestone and Kunga)  
483 averaged 0.97 visits/day/station, against 0.20 (five folds lower) for the deer native to Reef post-  
484 cull (Table 5.). Frequency of bait consumption per visit exceeded 75% on Kunga and East  
485 Limestone (no-risk) (Table 5.) and was 20% for animals native to after the culls on Reef. In the  
486 presence of bait, marked deer on East Limestone and Kunga consumed bait in over 80% of the  
487 events whereas unmarked deer consumed bait in 61% and 50% respectively of the visits. This is  
488 in striking contrast to native deer on Reef post-cull, where only three of the nine native deer  
489 trapped and marked post-cull were seen consuming bait at bait stations. Bait remained untouched  
490 by the 14 remaining (unmarked) Reef post-cull deer. Unmarked individuals, either briefly  
491 investigated the bait (visually and/or olfactory) without consuming it, or ignored it altogether,  
492 spending <30 s on site. When comparing median values, less affected by outliers, length of median  
493 time spent on a station exceeded 3 min for deer with no hunting history and was less than 30 s for  
494 post-cull deer on Reef.

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

507 In absence of bait because of previous consumption, mean visit length by deer at a station were  
508 shorter (Fig. 4d versus Fig. 4c). Sample still had a significant effect on length of visit at a station  
509 (log10 transformed): p = 0.05, F-statistic = 2.72 (on 3 predictor, DF=84), Residual standard error  
510 = 0.64, R<sup>2</sup> = 0.09 (Table 5, Fig. 4d), but this effect was unrelated to risk history. Visits were longer  
511 on average on Kunga essentially through the effect of outliers spending long periods at sites that  
512 had bait recently (Fig. 4d). Whatever the sample, time spent at a station never exceeded 4 minutes

513 (Table 5). Median values equaled 36 s or less, with the exception of Kunga deer (median = 84 s).  
 514 When time was spent at a station without bait, it was spent sniffing at the location where bait had  
 515 been present.

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

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

530

| 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   |                          |         |                          |          |        |

531

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

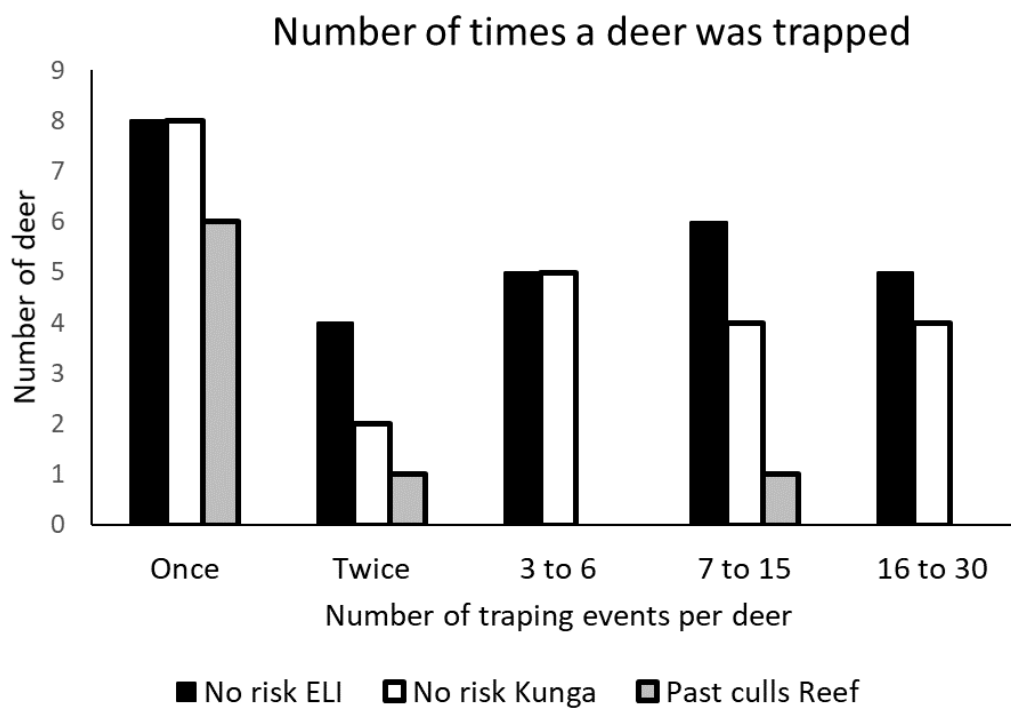
539 on Kunga, 15 out of 23 deer, were captured more than once. Even more, 13 of the deer captured  
 540 on each island were recaptured three times or more. In contrast, among the deer native to post-cull  
 541 Reef Island, only two of nine deer were captured more than once: one twice, and the other  
 542 individual repeatedly (Fig. 5).

543

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

| Island         | Years      | Effort<br>boxes * days | Nb. deer<br>captured | captures +<br>recaptures | Trapping success<br>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                             |

547



548

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

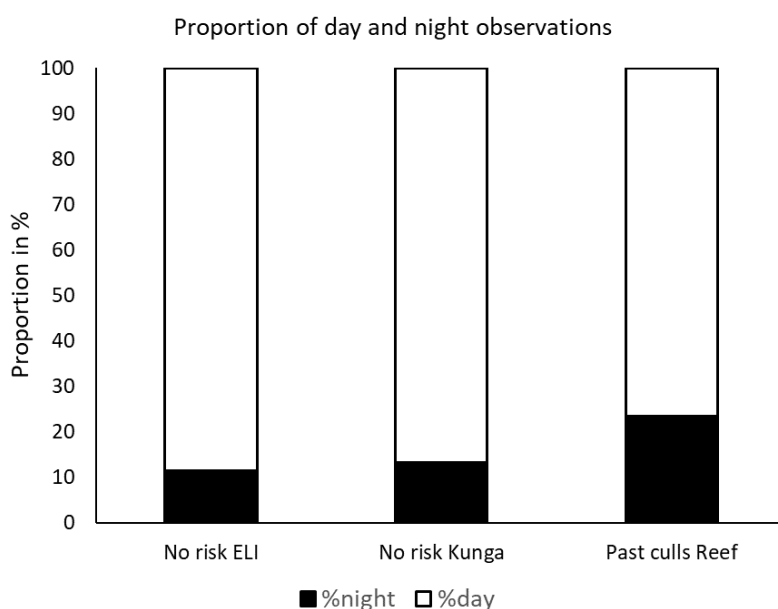
551 **Diel activity pattern varied with risk category**

552 Between 2011 and 2014 we recorded 762 (Table 7) picture sequences of deer on automatic  
 553 cameras. We used the time recorded on the first picture in a sequence to assign the sequence to  
 554 day time or night time. These pictures were taken in spring and early summer, hence a period of  
 555 the year of long days and short nights. After correcting for differences in duration of day and  
 556 night, deer were more active by day than by night (Fig. 6). There was no significant difference in  
 557 daily activity patterns between East Limestone and Kunga islands (no-risk) (Chi-squared = 0.29,  
 558 df = 1, p = 0.59) but deer native to post-cull Reef were more nocturnal than either (Chi-squared =  
 559 10.38, df = 1, p = 0.005). There was no significant difference between marked and unmarked deer  
 560 in diel activity either on Reef Island post-cull or on Kunga Island (Chi-squared = 0.05, df = 1, p =  
 561 0.81).

562 **Table 7.** Distribution of deer observations by automatic cameras across sites and their assignment to day  
 563 or night with day defined as the period between start of civil-twilight in the morning and end of civil-  
 564 twilight in the evening. Night was defined as the period between civil twilight end and civil twilight start  
 565 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%        |

566



**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)

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

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

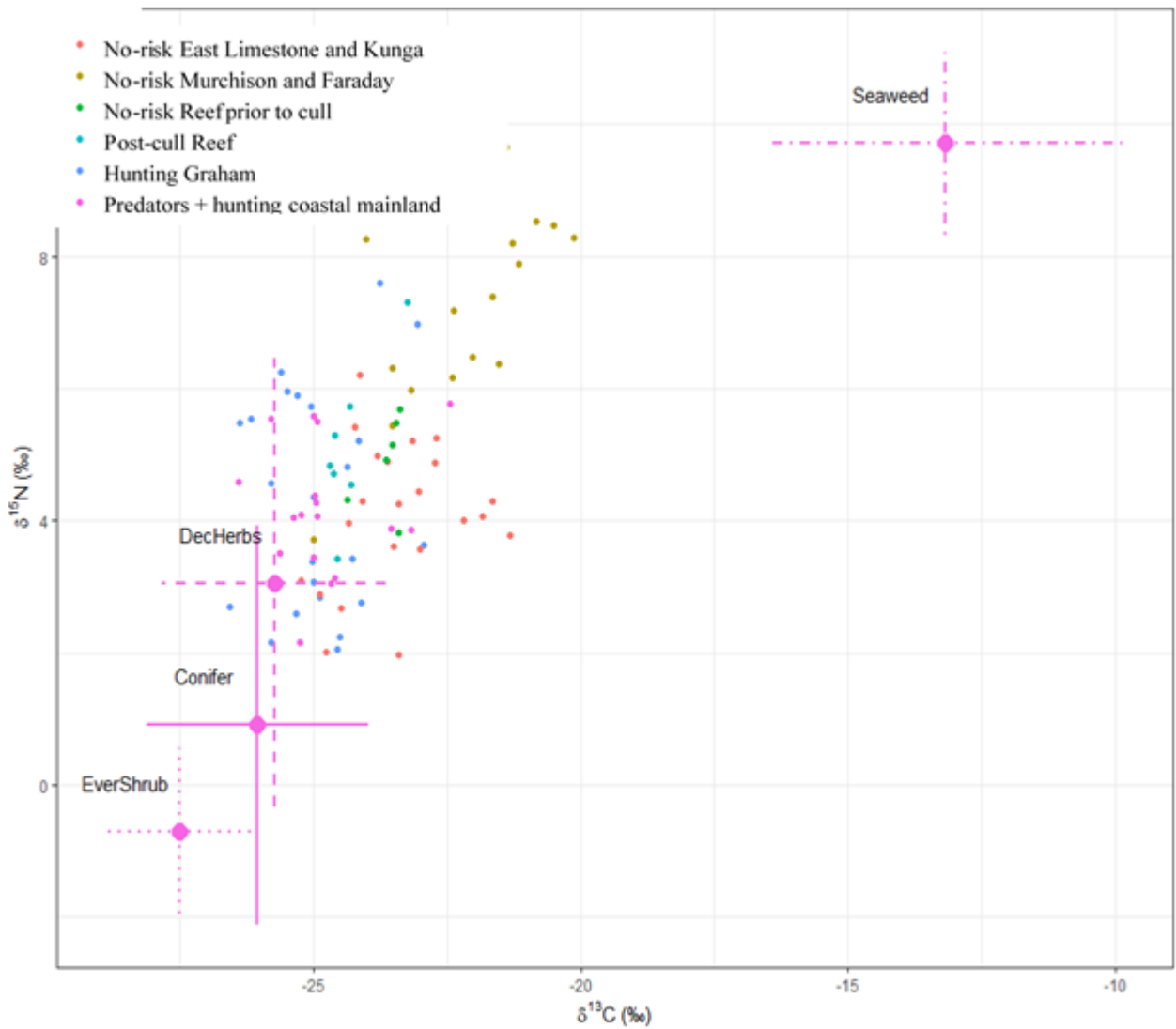
585 The isospace plot (Fig. 7) shows a spread of the isotopic signature in deer bone samples relative  
586 to the proportion of seaweed and terrestrial plants in deer diet. The samples with the highest  
587 proportion of seaweed-isotopes in jawbone collagen were from the “No-risk” islands Murchison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

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

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

604 In all situation the isotope data suggests that deer diets were dominated by the terrestrial plants  
605 group that included deciduous trees and shrubs, ferns, forbs and grasses. Its proportion varied from  
606 67% on “No-risk” Murchison and Faraday samples to 88% on “Hunting Graham” and in  
607 “Predators + hunting coastal mainland” samples. Conifers and evergreen shrubs stand out as a  
608 minor component of diet, despite their prevalence in the vegetation profiles of some of our study  
609 sites (see Fig. 3). The proportion of seaweeds in the “No-risk” Murchison and Faraday samples  
610 are remarkably high.





611

612 **Figure 7.** Isospace plots of deer and plants Mixture data (deer) by risk category and islands. No-risk  
 613 Murchison and Faraday = No-risk samples from Murchison (N=10 bone samples) and Faraday (N=6);  
 614 No-risk samples from East Limestone (N=10) and Kunga (N=12); No-risk samples from Reef Island  
 615 prior to the culls (N=7); Post-cull Reef = samples from Reef Island after the culls (N=6); Hunting  
 616 Graham = samples from Graham Island (N=23); Predators + hunting = samples from the coastal  
 617 mainland (N=17). Source data are by risk/island categories and have been adjusted by discrimination  
 618 means and SDs. Terrestrial plants, EverShrub = evergreen shrub, Conifer, DecHerbs = deciduous trees  
 619 and shrubs, forbs, ferns and grasses; Seaweed (40 species). Error bars indicate 1 SD, the combined  
 620 source+discrimination SD calculated under assumption of independence as:

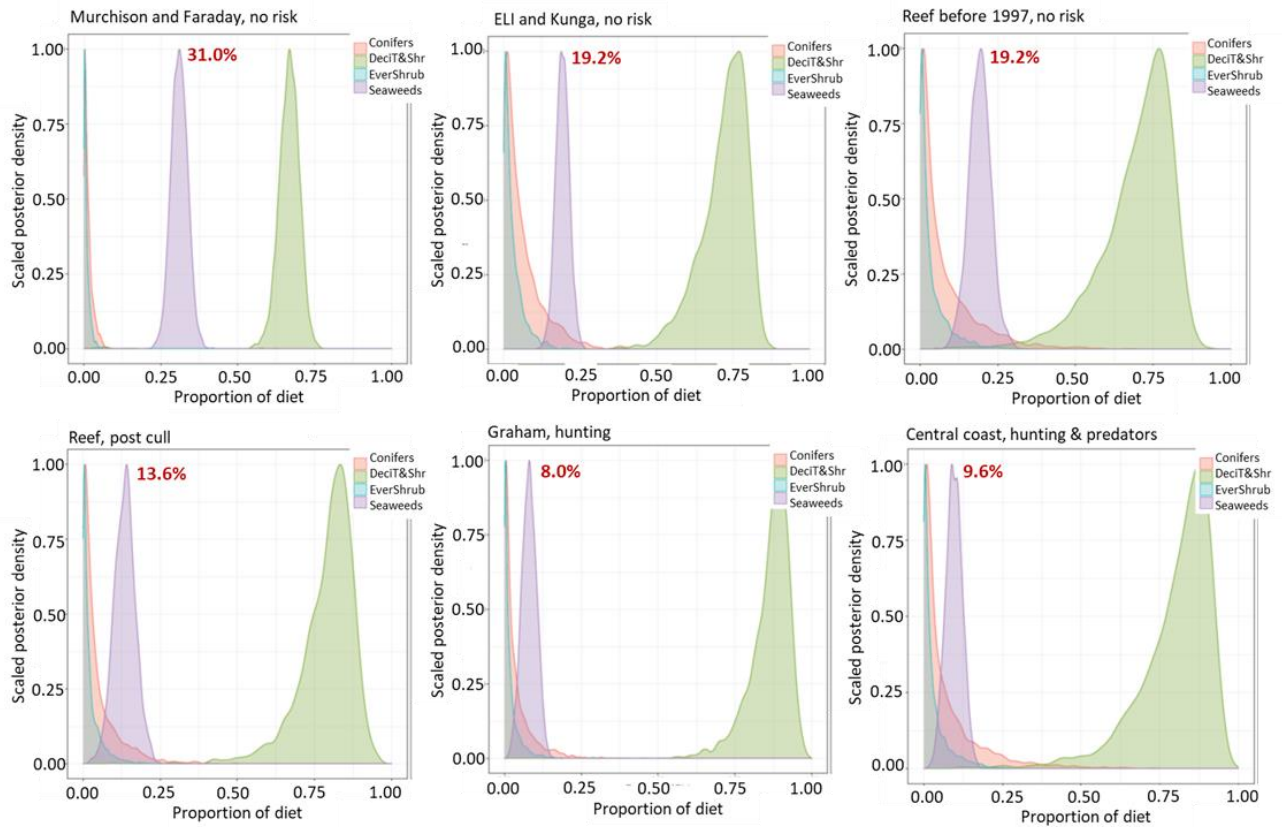
$$\sqrt{\sigma_{source}^2 + \sigma_{discr}^2}$$

621

622

623

624



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

637

638

639

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

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

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

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

670 Our comparison of anti-predator behaviours between populations that had not been exposed to  
671 hunting (East Limestone and Kunga Islands), with a population on Reef Island that had been  
672 subject, over a decade prior to the study, to a severe population reduction through experimental  
673 culling, revealed clear behavioural contrasts among these populations. Deer exposed to  
674 different risk history responded differently to cues mimicking predation risk. They also differed  
675 in their response to foreign food at bait stations, and to foreign food associated with an  
676 unfamiliar object. The significantly shorter flight initiation distances, and much shorter  
677 distances travelled, observed in non-hunted populations were retained by the Kunga deer  
678 translocated to Reef, even though they showed slightly longer flight initiation distances in their  
679 novel environment. This was in stark contrast with the wary behaviour of the deer native to  
680 post-cull Reef, even more so as the flight and travel distances we recorded for the post-cull  
681 Reef deer were likely conservative: they often fled before being seen (flight diagnosed by ear)  
682 and often were still running when they disappeared from view. This suggests that we  
683 underestimated distances travelled and that actual values on Reef may have been closer to  
684 values reported in black-tailed deer populations on the mainland (FID ~ 60 to 70 m and Dtravel  
685 ~ 70 to 120 m), where deer can be subject to hunting, poaching and carnivores (Stankowich  
686 and Coss 2006, 2007, Stankowich 2008). Our field notes also indicated that animals native to  
687 post-cull Reef ran or trotted away, whereas deer from Kunga and East Limestone (no-risk  
688 islands) usually performed a slow walk to a nearby location where they resumed their activities.

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

693 Despite minor differences in flight distances or time spent at bait, marked and unmarked deer  
694 of the same island had comparable behavioural profiles. In particular, marked and unmarked,  
695 deer from post-cull Reef Island were predominantly shy, as is illustrated by their poor trapping  
696 rate, even when we take into account that deer densities on post-cull Reef were still at only  
697 50% of their pre-cull density (~30 deer/km<sup>2</sup>, i.e. >80 deer present) (Chollet 2012). Only two  
698 (25%) of the eight deer captured on post-cull Reef were ever re-captured, and only one was  
699 recaptured repeatedly, mostly as a yearling and a juvenile, before becoming trap shy the  
700 following years while still captured on cameras. This contrasts with deer unexposed to risk on

701 East Limestone and Kunga, where over half of the deer trapped were re-trapped at least 3 times,  
702 often much more often over the years (Fig. 8).

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

709 Food limitation and nutritional status have also been proposed as mechanisms explaining  
710 individual attenuation in antipredator behaviours such as flight response to an approaching  
711 threat (Stankowich 2008, Gaynor et al. 2019). Such individual behavioural-adjustments,  
712 motivated by food shortage, should be lifted once food supply improves. This could explain  
713 the higher values of FID and Dtravel observed for Reef Island deer post-cull, as well as the less  
714 wary behaviour of deer living in heavily browsed understories. However, we did not observe  
715 any effect of food availability in the deer we translocated from Kunga to Reef. These animals  
716 did not change their behaviour despite a large increase in their available food supply. Our  
717 observations better fit the hypothesis that less wary behaviour was enabled by the absence of  
718 risk. Less wary behaviour could progressively favour individuals investing more in foraging  
719 and less in costly anti-predator behaviours. Less costly anti-predator behaviours such as routine  
720 vigilance would be retained and not counter-selected as observed for the populations of East  
721 Limestone and Kunga where routine vigilance persisted in an otherwise tame population.  
722 (Chamaillé-Jammes et al. 2014, Le Saout et al. 2015).

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

734 (Bonnot et al. 2016). But a higher proportion daytime activity in non-hunted deer populations  
735 on Haida Gwaii may also be interpreted as responding to the need for increased foraging time  
736 in intensively browsed understories.

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

738 Stockton et al. (2005) showed that, even on islands with the most severe deer impact on the  
739 understory within our study area, deer densities remained high, exceeding 20 deer per km<sup>2</sup>.  
740 However, Le Saout et al. (2014) showed that less than 5% of the resources needed by these  
741 dense deer populations were provided by the rare standing biomass. In summer the bulk of deer  
742 forage was provided equally by a cryptic flux of growing vegetation and by canopy fall in the  
743 form of foliage shed by wind. In winter canopy fall was the dominant source of forage  
744 supplemented by seaweed available on the shorelines (Le Saout et al. 2014). But these  
745 resources, while sufficient, occurred in small spatially-dispersed cryptic items in contrast with  
746 the volume and distribution of palatable foliage and plant material in lush understories on  
747 islands without deer (Martin et al. 2010) or with limited deer impact as on Yeo (Fig. 3).  
748 Spatially dispersed food occurring as small items, could favour behaviour that increases  
749 foraging time at the expense of time devoted to costly anti-predator behaviours (Réale et al.  
750 2010, Sih et al. 2012) such as flight or dynamic adjustments of spatial distribution in response  
751 to perceived distribution of risk (Williams et al. 2008, Gaynor et al. 2019). This could favour  
752 a progressive decrease in understory vegetation, as was actually shown on these islands by  
753 (Chollet et al. 2015) between 1989 and 2009.

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

755 Cursorial predator hunting by sight should increase the reluctance in their prey to use open  
756 habitats (Bonnot et al. 2017). In addition, the lush understory on Yeo Island, similar in cover  
757 and diversity to the understory of islands never colonized by deer, or the lush understory on  
758 post-cull Reef Island, may reduce the incentive to use the shoreline in these deer. The data on  
759 understory vegetation (Fig. 3), also suggest a potential for a negative relationship between the  
760 amount of vegetation in the understory and the proportion of seaweed signature in deer bone  
761 collagen. The very high seaweed isotopic signature on “No-risk” Murchison and Faraday,  
762 could, for instance, be interpreted as the consequences of a lack of understory forage on these  
763 two islands. The vegetation data collected in 20 plots on “No-risk” Murchison Island in 1993  
764 (Martin et al. 1995) indeed recorded less than 4% of ligneous and herbaceous vegetation cover  
765 in the 0 to 1.5 m layer, making it one of the least vegetated understories in the archipelago [see

766 Fig. 2 in Martin et al. (1995), an observation confirmed by more recent visits to Murchison,  
767 also see Chollet et al. (2015)]. In contrast to the other islands, Murchison and Faraday islands  
768 had been subjected to extensive logging in the first half of the 20th century, resulting in a dense  
769 canopy, further reducing forage for deer. Such a high proportion of seaweed in ruminant diet  
770 is not unheard of, in particular in conditions where other resources were scarce or unavailable.  
771 Seaweeds accounted for up to 50, and even 100%, of the diet of some sheep in primitive North  
772 Ronaldsay (Orkey islands) sheep (Balasse et al. 2005) that were confined to the shoreline.

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

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

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

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

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

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

819 Our results are consistent with the hypothesis that changes in the landscape of fear (Gaynor et  
820 al. 2019, Palmer et al. 2022) can alter the behaviour characteristic of a population. Such  
821 changes can result from severe differential selection of certain behavioural syndromes driven  
822 by acute risk (e.g. as in the case of severe culling), or, over longer periods of time, from the  
823 progressive reproductive advantage of behavioural profiles better adapted to a changing  
824 environment (e.g. driven by a progressive depletion of resources in absence of acute predation  
825 risk). Although this remains speculative in the case of the non-wary behaviours we observed  
826 on islands without predation or hunting, it illustrates a potential for long term selection. The  
827 rapidity of adaptive behavioural change will depend on the magnitude of the selective pressure



828 (Blumstein 2002, Jolly et al. 2018, Moseby et al. 2023). It can be rapid if severe culling is  
829 maintained over generations continuously but will presumably be slower in the case of the  
830 relaxation of antipredator behaviours after the colonisation of a predator free environment  
831 (Blumstein 2002). The evolution of predator naivety in island organisms provides numerous  
832 text-book examples of long-term progressive (or fast) evolutionary shifts in behavioural traits  
833 (Darwin 1840, Lack 1968, Blumstein 2002), but see Blumstein and Daniel (2005).

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

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

849

850

#### **Ethical note**

851 All research was conducted under the appropriate animal care permits from the Wildlife Act  
852 of the Ministry of Natural Resources Operation of British Columbia (No. NA11-68421,  
853 approved by Parks Canada Animal Care Task Force research permit No. 9059), and under the  
854 Archipelago Management Board of Gwaii Haanas National Park Reserve and Haida Heritage  
855 Site research permit No. GWA-2011-8373. All except 3 deer captures were done using box  
856 traps specially designed for this project (the exceptions were one individual captured by a  
857 clover trap and two by a netgun). We made over 400 deer captures or recaptures in the course  
858 of the project. Traps were radio-monitored remotely and we dealt with captured animal  
859 immediately after capture notification. The animals captured at night were treated the following

860 morning. We recorded no injury to deer, nor mortality associated to capture. The project  
861 permits included the experimental culls on one island and the translocation of a small number  
862 of deer from one island to another. Culls were conducted by local hunters under clear ethical  
863 and practical rules. Carcasses were processed and the meat distributed to the local  
864 communities.

### 865 **Acknowledgements**

866 This is a publication of the Research Group on Introduced Species (RGIS,  
867 <https://rgis.cefe.cnrs.fr> ). On top of funding (see below) we received critical logistic support  
868 from the Laskeek Bay Conservation Society, Environment and Climate Change Canada, the  
869 Gwaii Haanas National Park Reserve, National Marine Conservation Area Reserve and Haida  
870 Heritage Site, the Coast Sustainability Trust Matching Fund (CST-QCI-HG- 061– RGIS  
871 Infrastructure Upgrade), and the “Understanding Canada” program from the Canadian  
872 government. We are indebted to Todd Golumbia, Carita Bergman and Peter Dymont and  
873 colleagues from Gwaii Haanas for support in the field. Tibo Verchère, Ainsley Brown, Barbara  
874 and Keith Rowsell, Greg Martin, Jacques Morin, Erin Harris, Terry and Ron Husband, Bruno  
875 Cargnelutti, Nicolas Cebe, Mike Gillingham, Malcolm Hyatt, Georges Janeau, Jean-Luc  
876 Rames, Denis Picot, Helen Schwantje, Tom Smith and Keith Tipper provided critical help in  
877 the field and beyond. Chris Darimont, Eva Poilvé and Maurah Van Impe assisted with the  
878 treatment of samples in the stable isotope analysis. The Haida Watchmen program provided  
879 precious help through accommodation on Tanu Island. Over the years, many members of the  
880 local communities provided invaluable support. We are deeply thankful to all.

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

882 N/A at this stage

### 883 **Funding**

884 Funding for this research included grants from the French Ministry of Foreign Affairs, from  
885 the French National Agency for Research (#2010-BLAN-1718, BAMBI), the International  
886 Research Group (GDRI) Dynamics of Biodiversity and Life-History traits from CNRS, the  
887 South Moresby Forest Replacement Account (SMFRA) and Gwaii Forestry Charitable Trust  
888 (GF), and from National Geographic # 9266-13.

889

890

### Conflict of interest disclosure

891 The authors declare that they comply with the PCI rule of having no financial conflicts of  
892 interest in relation to the content of the article. *JL Martin and Simon Chamaille-Jammes are*  
893 *listed as recommenders*

894

### 895 References

896 Altendorf, K. B., Landré, J. W., López González, C. A., & Brown, J. S. (2001). Assessing  
897 effects of predation risk on foraging behavior of mule deer. *Journal of Mammalogy*, 82(2),  
898 430–439.

899 Balasse, M., Tresset, A., Dobney, K., & Ambrose, S. H. (2005). The use of isotope ratios to  
900 test for seaweed eating in sheep. *Journal of Zoology*, 266(3), 283–291.

901 Ballard, W. B., Lutz, D., Keegan, T. W., Carpenter, L. H., & deVos Jr, J. C. (2001). Deer-  
902 predator relationships: A review of recent North American studies with emphasis on mule and  
903 black-tailed deer. *Wildlife Society Bulletin*, 99–115.

904 Banner, A., MacKenzie, W. H., Pojar, J., MacKinnon, A., Saunders, S. C., & Klassen, H.  
905 (2014). A field guide to ecosystem classification and identification for Haida Gwaii. *Land*  
906 *Management Handbook*.

907 Blumstein, D. T. (2002). Moving to suburbia: Ontogenetic and evolutionary consequences of  
908 life on predator-free islands. *Journal of Biogeography*, 29(5-6), 685–692.

909 Blumstein, D. T., & Daniel, J. C. (2005). The loss of anti-predator behaviour following  
910 isolation on islands. *Proceedings of the Royal Society B: Biological Sciences*, 272(1573),  
911 1663–1668.

912 Bocherens, H. (1999). Isotopes stables et reconstitution du régime alimentaire des hominidés  
913 fossiles: Une revue. *Bulletins et Mémoires de La Société d'Anthropologie de Paris*, 11(3), 261–  
914 287.

915 Bocherens, H., & Drucker, D. (2003). Trophic level isotopic enrichment of carbon and nitrogen  
916 in bone collagen: Case studies from recent and ancient terrestrial ecosystems. *International*  
917 *Journal of Osteoarchaeology*, 13(1-2), 46–53.

918 Bocherens, H., Fizet, M., Cuif, J.-P., Jaeger, J.-J., & Michard, J.-G. (1988). Premières mesures  
919 d'abondance isotopiques naturelles en  $^{13}\text{C}$  et  $^{15}\text{N}$  de la matière organique fossile de

920 Dinosaur. Application à l'étude du régime alimentaire du genre Anatosaurus (Ornithischia,  
921 Hadrosauridae). *Comptes Rendus de l'Académie Des Sciences. Série 2, Mécanique, Physique,*  
922 *Chimie, Sciences de l'univers, Sciences de La Terre*, 306(20), 1521–1525.

923 Bochéréns, Hervé, Marc Fizet, Lange-Badre, B., Vandermeersch, B., Borel, J. P., & Bellon, G.  
924 (1991). Isotopic biogeochemistry ( $^{13}\text{C}$ ,  $^{15}\text{N}$ ) of fossil vertebrate collagen: Application to the  
925 study of a past food web including Neandertal man. *Journal of Human Evolution*, 20(6), 481–  
926 492.

927 Bøhn, T., & Amundsen, P.-A. (2004). Ecological interactions and evolution: Forgotten parts  
928 of biodiversity? *BioScience*, 54(9), 804–805.

929 Bonnot, N. C., Couriot, O., Berger, A., Cagnacci, F., Ciuti, S., De Groeve, J. E., Gehr, B.,  
930 Heurich, M., Kjellander, P., & Kröschel, M. (2020). Fear of the dark? Contrasting impacts of  
931 humans versus lynx on diel activity of roe deer across Europe. *Journal of Animal Ecology*,  
932 89(1), 132–145.

933 Bonnot, N. C., Hewison, A. M., Morellet, N., Gaillard, J.-M., Debeffe, L., Couriot, O.,  
934 Cargnelutti, B., Chaval, Y., Lourtet, B., & Kjellander, P. (2017). Stick or twist: Roe deer adjust  
935 their flight behaviour to the perceived trade-off between risk and reward. *Animal Behaviour*,  
936 124, 35–46.

937 Bonnot, N. C., Morellet, N., Hewison, A. M., Martin, J.-L., Benhamou, S., & Chamaillé-  
938 Jammes, S. (2016). Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) adjust habitat  
939 selection and activity rhythm to the absence of predators. *Canadian Journal of Zoology*, 94(6),  
940 385–394.

941 Brown, J. S., Laundré, J. W., & Gurung, M. (1999). The ecology of fear: Optimal foraging,  
942 game theory, and trophic interactions. *Journal of Mammalogy*, 80(2), 385–399.

943 Burgess, B. T., Irvine, R. L., Martin, J.-L., & Russello, M. A. (2023). Past population control  
944 biases interpretations of contemporary genetic data: Implications for future invasive Sitka  
945 black-tailed deer management in Haida Gwaii. *Biological Invasions*, 25(12), 3871–3886.

946 Burgess, B. T., Irvine, R. L., & Russello, M. A. (2022a). A genotyping-in-thousands by  
947 sequencing panel to inform invasive deer management using noninvasive fecal and hair  
948 samples. *Ecology and Evolution*, 12(6), e8993.

949 Burgess, B. T., Irvine, R. L., & Russello, M. A. (2022b). Population genomics of Sitka black-  
950 tailed deer supports invasive species management and ecological restoration on islands.  
951 *Communications Biology*, 5(1), 223.

952 Callan, R., Nibbelink, N. P., Rooney, T. P., Wiedenhoeft, J. E., & Wydeven, A. P. (2013).  
953 Recolonizing wolves trigger a trophic cascade in Wisconsin (USA). *Journal of Ecology*,  
954 101(4), 837–845. <https://doi.org/10.1111/1365-2745.12095>

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

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

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

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

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

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

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

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

976 Chollet, S., Bergman, C., Gaston, A. J., & Martin, J.-L. (2015). Long-term consequences of  
977 invasive deer on songbird communities: Going from bad to worse? *Biological Invasions*, 17(2),  
978 777–790.

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

982 Chollet, S., Padié, S., Stockton, S., Allombert, S., Gaston, A. J., & Martin, J.-L. (2016a).  
983 Positive plant and bird diversity response to experimental deer population reduction after  
984 decades of uncontrolled browsing. *Diversity and Distributions*, *22*(3), 274–287.

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

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

991 Côté, S. D., Rooney, T. P., Tremblay, J.-P., Dussault, C., & Waller, D. M. (2004). Ecological  
992 Impacts of Deer Overabundance. *Annual Review of Ecology, Evolution, and Systematics*, *35*,  
993 113–147.

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

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

998 Darwin, C. (1840). *Journal of Researches into the Geology and Natural History of the Varoius*  
999 *Countries Visited by HMS Beagle, under the Command of Captain Fitzroy from 1832 to 1836*  
1000 *by Charles Darwin*. Colburn.

1001 Daufresne, T., & Martin, J. L. (1997). Changes in vegetation structure and diversity as a result  
1002 of browsing by a large herbivore: The impact of introduced Black-tailed deer in the primary  
1003 forest of Haida Gwaii, British Columbia. *Laskeek Bay Research*, *7*, 2–26.

1004 DeNiro, M. J., & Epstein, S. (1978). Influence of diet on the distribution of carbon isotopes in  
1005 animals. *Geochimica et Cosmochimica Acta*, *42*(5), 495–506.

1006 DeNiro, M. J., & Epstein, S. (1981). Influence of diet on the distribution of nitrogen isotopes  
1007 in animals. *Geochimica et Cosmochimica Acta*, *45*(3), 341–351.

1008 Ellers, J., Toby Kiers, E., Currie, C. R., McDonald, B. R., & Visser, B. (2012). Ecological  
1009 interactions drive evolutionary loss of traits. *Ecology Letters*, *15*(10), 1071–1082.

1010 Elton, C. (1927). *Animal ecology*. 207 pp. Sidgwick & Jackson, LTD. London.

1011 Engelstoft, C. (2001). Effects of Sitka Black-tailed Deer (*Odocoileus Hemionus Sitkensis*) on  
1012 Understory in Old Growth Forest on Haida Gwaii (Queen Charlotte Islands), British Columbia.  
1013 University of Victoria.

1014 Engelstoft, C., Kirchhoff, M. D., & Eastman, D. (2008). Herbivory and the missing understory  
1015 on Haida Gwaii. In *Lessons from the Islands: Introduced Species and What They Tell Us About*  
1016 *How Ecosystems Work*. (pp. 78–86). Canadian Wildlife Service, Environment Canada, Ottawa.

1017 Estes, J. A., Brashares, J. S., & Power, M. E. (2013). Predicting and detecting reciprocity  
1018 between indirect ecological interactions and evolution. *The American Naturalist*, *181*(S1),  
1019 S76–S99.

1020 Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., Carpenter, S.  
1021 R., Essington, T. E., Holt, R. D., Jackson, J. B. C., Marquis, R. J., Oksanen, L., Oksanen, T.,  
1022 Paine, R. T., Pikitch, E. K., Ripple, W. J., Sandin, S. A., Scheffer, M., Schoener, T. W., ...  
1023 Wardle, D. A. (2011). Trophic Downgrading of Planet Earth. *Science*, *333*(6040), 301–306.

1024 Gaston, A. J., Stockton, S. A., & Smith, J. L. (2006). Species-area relationships and the impact  
1025 of deer-browse in the complex phylogeography of the Haida Gwaii archipelago (Queen  
1026 Charlotte Islands), British Columbia. *Écoscience*, *13*(4), 511–522.  
1027 [https://doi.org/10.2980/1195-6860\(2006\)13\[511:SRATIO\]2.0.CO;2](https://doi.org/10.2980/1195-6860(2006)13[511:SRATIO]2.0.CO;2)

1028 Gaynor, K. M., Brown, J. S., Middleton, A. D., Power, M. E., & Brashares, J. S. (2019).  
1029 Landscapes of fear: Spatial patterns of risk perception and response. *Trends in Ecology &*  
1030 *Evolution*, *34*(4), 355–368.

1031 Golumbia, T., Bland, L., Moore, K., & Bartier, P. (2008). History and current status of  
1032 introduced vertebrates on Haida Gwaii. In *Lessons from the islands: Introduced species and*  
1033 *what they tell us about how ecosystems work* (pp. 8–31). Canadian Wildlife Service,  
1034 Environment Canada, Ottawa.

1035 Greenberg, R., & Mettke-Hofmann, C. (2001). Ecological aspects of neophobia and neophilia  
1036 in birds. *Current Ornithology*, 119–178.

- 1037 Haeckel, E. (1866). *Generelle Morphologie der Organismen. Allgemeine Grundzüge der*  
1038 *organischen Formen-Wissenschaft, mechanisch begründet durch die von C. Darwin reformirte*  
1039 *Descendenz-Theorie. Allgemeine Entwicklungsgeschichte. (Vol. 2).* De Gruyter.
- 1040 Janzen, D. H. (1974). The deflowering of central america. *Natural History Magazine*, 83(4),  
1041 48–51.
- 1042 Jolly, C. J., Webb, J. K., & Phillips, B. L. (2018). The perils of paradise: An endangered species  
1043 conserved on an island loses antipredator behaviours within 13 generations. *Biology Letters*,  
1044 14(6), 20180222. <https://doi.org/10.1098/rsbl.2018.0222>
- 1045 Kelly, J. F. (2000). Stable isotopes of carbon and nitrogen in the study of avian and mammalian  
1046 trophic ecology. *Canadian Journal of Zoology*, 78(1), 1–27.
- 1047 Klein, D. R. (1995). The introduction, increase, and demise of wolves on Coronation Island,  
1048 Alaska. *Ecology and Conservation of Wolves in a Changing World*, 275, 280.
- 1049 Kuijper, D. P. J., Sahlén, E., Elmhagen, B., Chamaillé-Jammes, S., Sand, H., Lone, K., &  
1050 Cromsigt, J. (2016). Paws without claws? Ecological effects of large carnivores in  
1051 anthropogenic landscapes. *Proc. R. Soc. B*, 283. <http://dx.doi.org/10.1098/rspb.2016.162>
- 1052 Lack, D. L. (1968). *Ecological adaptations for breeding in birds.* Methuen.
- 1053 Le Saout, S., Chollet, S., Chamaillé-Jammes, S., Blanc, L., Padié, S., Verchere, T., Gaston, A.  
1054 J., Gillingham, M. P., Gimenez, O., Parker, K. L., Picot, Verheyden, & Martin, J.-L. (2014).  
1055 Understanding the paradox of deer persisting at high abundance in heavily browsed habitats.  
1056 *Wildlife Biology*, 20(3), 122–135.
- 1057 Le Saout, S., Martin, J.-L., Blanchard, P., Cebe, N., Mark Hewison, A. J., Rames, J.-L., &  
1058 Chamaillé-Jammes, S. (2015). Seeing a ghost? Vigilance and its drivers in a predator-free  
1059 world. *Ethology*, 121(7), 651–660.
- 1060 Lima, S. L., & Dill, L. M. (1990). Behavioral decisions made under the risk of predation: A  
1061 review and prospectus. *Canadian Journal of Zoology*, 68(4), 619–640.
- 1062 Maillard, M., Martin, J.-L., Chollet, S., Catomeris, C., Simon, L., & Grayston, S. J. (2021).  
1063 Belowground effects of deer in a temperate forest are time-dependent. *Forest Ecology and*  
1064 *Management*, 493, 119228.



- 1065 Martin, J.-L., & Baltzinger, C. (2002a). Interaction among deer browsing, hunting, and tree  
1066 regeneration. *Canadian Journal of Forest Research*, 32(7), 1254–1264.  
1067 <https://doi.org/10.1139/x02-043>
- 1068 Martin, J.-L., & Baltzinger, C. (2002b). Interaction among deer browsing, hunting, and tree  
1069 regeneration. *Canadian Journal of Forest Research*, 32(7), 1254–1264.
- 1070 Martin, J.-L., Chamaille-Jammes, S., & Waller, D. M. (2020). Deer, wolves, and people: Costs,  
1071 benefits and challenges of living together. *Biological Reviews*, 93(3), 782–801.
- 1072 Martin, J.-L., Gaston, A. J., & Hitier, S. (1995). The effect of island size and isolation on old  
1073 growth forest habitat and bird diversity in Gwaii Haanas (Queen Charlotte Islands, Canada).  
1074 *Oikos*, 115–131.
- 1075 Martin, J.-L., Stockton, S. A., Allombert, S., & Gaston, A. J. (2010a). Top-down and bottom-  
1076 up consequences of unchecked ungulate browsing on plant and animal diversity in temperate  
1077 forests: Lessons from a deer introduction. *Biological Invasions*, 12(2), 353–371.  
1078 <https://doi.org/10.1007/s10530-009-9628-8>
- 1079 Martin, J.-L., Stockton, S. A., Allombert, S., & Gaston, A. J. (2010b). Top-down and bottom-  
1080 up consequences of unchecked ungulate browsing on plant and animal diversity in temperate  
1081 forests: Lessons from a deer introduction. *Biological Invasions*, 12(2), 353–371.  
1082 <https://doi.org/10.1007/s10530-009-9628-8>
- 1083 Martin, T. G., Arcese, P., & Scheerder, N. (2011). Browsing down our natural heritage: Deer  
1084 impacts on vegetation structure and songbird populations across an island archipelago.  
1085 *Biological Conservation*, 144(1), 459–469.
- 1086 Mathews, N. E., & Porter, W. F. (1988). Black bear predation of white-tailed deer neonates in  
1087 the central Adirondacks. *Canadian Journal of Zoology*, 66(5), 1241–1242.
- 1088 Monestier, C., Morellet, N., Verheyden, H., Gaillard, J.-M., Bideau, E., Denailhac, A., Lourtet,  
1089 B., Cebe, N., Picot, D., & Rames, J.-L. (2017). Neophobia is linked to behavioural and  
1090 haematological indicators of stress in captive roe deer. *Animal Behaviour*, 126, 135–143.
- 1091 Moseby, K. E., Blumstein, D. T., & Letnic, M. (2016). Harnessing natural selection to tackle  
1092 the problem of prey naïveté. *Evolutionary Applications*, 9(2), 334–343.
- 1093 Moseby, K. E., Carthey, A., & Schroeder, T. (2015). The influence of predators and prey  
1094 naivety on reintroduction success: Current and future directions. *Advances in Reintroduction  
1095 Biology of Australian and New Zealand Fauna*, 29–42.

1096 Moseby, K., Van der Weyde, L., Letnic, M., Blumstein, D. T., West, R., & Bannister, H.  
1097 (2023). Addressing prey naivety in native mammals by accelerating selection for antipredator  
1098 traits. *Ecological Applications*, 33(2), e2780.

1099 Paine, R. T. (1966). Food web complexity and species diversity. *The American Naturalist*,  
1100 100(910), 65–75.

1101 Paine, R. T. (1969). A Note on Trophic Complexity and Community Stability. *The American*  
1102 *Naturalist*, 103(929), 91–93.

1103 Palmer, M. S., Gaynor, K. M., Becker, J. A., Abraham, J. O., Mumma, M. A., & Pringle, R.  
1104 M. (2022). Dynamic landscapes of fear: Understanding spatiotemporal risk. *Trends in Ecology*  
1105 *& Evolution*, 37(10). <https://doi.org/10.1016/j.tree.2022.06.007>

1106 Pojar, J. (1999). The effects of deer browsing on the plant life of Haida Gwaii. Proceedings of  
1107 the Cedar Symposium: Growing Western Redcedar and Yellowcypress on the Queen Charlotte  
1108 Islands/Haida Gwaii. BC Ministry of Forests/Canada–British Columbia South Moresby Forest  
1109 Replacement Account, Victoria, BC, 90–97.

1110 Racine, J. S. (2012). RStudio: A platform-independent IDE for R and Sweave. JSTOR.

1111 Réale, D., Garant, D., Humphries, M. M., Bergeron, P., Careau, V., & Montiglio, P.-O. (2010).  
1112 Personality and the emergence of the pace-of-life syndrome concept at the population level.  
1113 *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1560), 4051–  
1114 4063.

1115 Réale, D., Reader, S. M., Sol, D., McDougall, P. T., & Dingemanse, N. J. (2007). Integrating  
1116 animal temperament within ecology and evolution. *Biological Reviews*, 82(2), 291–318.

1117 Richards, M. P., Fuller, B. T., & Molleson, T. I. (2006). Stable isotope palaeodietary study of  
1118 humans and fauna from the multi-period (Iron Age, Viking and Late Medieval) site of Newark  
1119 Bay, Orkney. *Journal of Archaeological Science*, 33(1), 122–131.

1120 Rodriguez-Prieto, I., Fernández-Juricic, E., Martín, J., & Regis, Y. (2009). Antipredator  
1121 behavior in blackbirds: Habituation complements risk allocation. *Behavioral Ecology*, 20(2),  
1122 371–377.

1123 Roffler, G. H., Eriksson, C. E., Allen, J. M., & Levi, T. (2023). Recovery of a marine keystone  
1124 predator transforms terrestrial predator–prey dynamics. *Proceedings of the National Academy*  
1125 *of Sciences*, 120(5), e2209037120.

1126 Salomon, A. K., Shears, N. T., Langlois, T. J., & Babcock, R. C. (2008). Cascading effects of  
1127 fishing can alter carbon flow through a temperate coastal ecosystem. *Ecological Applications*,  
1128 *18*(8), 1874–1887.

1129 Schulting, R. J., Blockley, S. M., Bocherens, H., Drucker, D., & Richards, M. (2008). Stable  
1130 carbon and nitrogen isotope analysis on human remains from the Early Mesolithic site of La  
1131 Vergne (Charente-Maritime, France). *Journal of Archaeological Science*, *35*(3), 763–772.

1132 Schuttler, S. G., Parsons, A. W., Forrester, T. D., Baker, M. C., McShea, W. J., Costello, R., &  
1133 Kays, R. (2017). Deer on the lookout: How hunting, hiking and coyotes affect white-tailed deer  
1134 vigilance. *Journal of Zoology*, *301*(4), 320–327.

1135 Sih, A., Bell, A., & Johnson, J. C. (2004). Behavioral syndromes: An ecological and  
1136 evolutionary overview. *Trends in Ecology & Evolution*, *19*(7), 372–378.

1137 Sih, A., Cote, J., Evans, M., Fogarty, S., & Pruitt, J. (2012). Ecological implications of  
1138 behavioural syndromes. *Ecology Letters*, *15*(3), 278–289.

1139 Soulé, M. E., Estes, J. A., Berger, J., & Del Rio, C. M. (2003). Ecological effectiveness:  
1140 Conservation goals for interactive species. *Conservation Biology*, *17*(5), 1238–1250.

1141 Stankowich, T. (2008). Ungulate flight responses to human disturbance: A review and meta-  
1142 analysis. *Biological Conservation*, *141*(9), 2159–2173.

1143 Stankowich, T., & Coss, R. G. (2006). Effects of predator behavior and proximity on risk  
1144 assessment by Columbian black-tailed deer. *Behavioral Ecology*, *17*(2), 246–254.

1145 Stankowich, T., & Coss, R. G. (2007). Effects of risk assessment, predator behavior, and  
1146 habitat on escape behavior in Columbian black-tailed deer. *Behavioral Ecology*, *18*(2), 358–  
1147 367.

1148 Stock, B. C., Jackson, A. L., Ward, E. J., Parnell, A. C., Phillips, D. L., & Semmens, B. X.  
1149 (2018). Analyzing mixing systems using a new generation of Bayesian tracer mixing models.  
1150 *PeerJ*, *6*, e5096.

1151 Stock, B., Semmens, B., Ward, E., Parnell, A., Jackson, A., & Phillips, D. (2018). Package  
1152 ‘MixSIAR.’ *Bayesian Mixing Models in R, Version*, *3*(10).

1153 Stockton, S. A., Allombert, S., Gaston, A. J., & Martin, J.-L. (2005). A natural experiment on  
1154 the effects of high deer densities on the native flora of coastal temperate rain forests. *Biological*  
1155 *Conservation*, *126*(1), 118–128. <https://doi.org/10.1016/j.biocon.2005.06.006>

1156 Taylor, W. P. (1956). *The Deer of North America*. Stackpole Company and Wildlife  
1157 Management Institute, Washington.

1158 Team, R. C. (2018). R: A language and environment for statistical computing; 2018.

1159 Valiente-Banuet, A., Aizen, M. A., Alcántara, J. M., Arroyo, J., Cocucci, A., Galetti, M.,  
1160 García, M. B., García, D., Gómez, J. M., & Jordano, P. (2015). Beyond species loss: The  
1161 extinction of ecological interactions in a changing world. *Functional Ecology*, 29(3), 299–307.

1162 Vila, B., Torre, F., Guibal, F., & Martin, J.-L. (2004). Can we reconstruct browsing history and  
1163 how far back? Lessons from *Vaccinium parvifolium* Smith in Rees. *Forest Ecology and*  
1164 *Management*, 201(2), 171–185.

1165 Waller, D. M., & Reo, N. J. (2018). First stewards: Ecological outcomes of forest and wildlife  
1166 stewardship by indigenous peoples of Wisconsin, USA. *Ecology and Society*, 23(1), 45.  
1167 <https://doi.org/10.5751/ES-09865-230145>

1168 Williams, S. C., DeNicola, A. J., & Ortega, I. M. (2008). Behavioral responses of white-tailed  
1169 deer subjected to lethal management. *Canadian Journal of Zoology*, 86(12), 1358–1366.

1170 Ydenberg, R. C., & Dill, L. M. (1986). The economics of fleeing from predators. *Advances in*  
1171 *the Study of Behavior*, 16, 229–249.

1172