

# 1 **Managing beach access and vehicle impacts following** 2 **reconfiguration of the landscape by a natural hazard event**

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## 11 12 **Abstract**

13 After New Zealand's 7.8 Mw Kaikōura earthquake in late 2016 an unexpected anthropogenic effect  
14 involved increased motorised vehicle access to beaches. We show how these effects were generated  
15 by landscape reconfiguration associated with coastal uplift and widening of high-tide beaches, and  
16 present analyses of the distribution of natural environment values in relation to vehicle movements  
17 and impacts. Access changes led to extensive vehicle tracking in remote areas that had previously  
18 been protected by natural barriers. New dunes formed seaward of old dunes and have statutory  
19 protection as threatened ecosystems, yet are affected by vehicle traffic. Nesting grounds of  
20 nationally vulnerable banded dotterel (*Charadrius bicinctus bicinctus*) co-occur with vehicle tracking.  
21 An artificial nest experiment showed that vehicle strikes pose risks to nesting success, with 91% and  
22 83% of nests destroyed in high and moderate-traffic areas, respectively, despite an increase in  
23 suitable habitat. Despite gains for recreational vehicle users there are serious trade-offs with  
24 environmental values subject to legal protection and associated responsibilities for management  
25 authorities. In theory, a combination of low-impact vehicle access and environmental protection  
26 could generate win-win outcomes from the landscape changes, but is difficult to achieve in practice.  
27 Detailed information on sensitive areas would be required to inform designated vehicle routes as a  
28 potential solution, and such sensitivities are widespread. Alternatively, vehicle access areas that  
29 accommodate longstanding activities such as boat launching could be formally established using  
30 identified boundaries to control impacts further afield. Difficulties for the enforcement of regulatory  
31 measures in remote areas also suggest a need for motivational strategies that incentivise low-impact  
32 behaviours. We discuss options for user groups to voluntarily reduce their impacts, the importance  
33 of interactions at the recreation-conservation nexus, and need for timely impact assessments across

34 the social-ecological spectrum after physical environment changes -- all highly transferable principles  
35 for other natural hazard and disaster recovery settings worldwide.

36 **Keywords:** nature-based recreation, recreational access, shoreline management, impact assessment,  
37 disaster recovery, protected areas

## 38 **1. Introduction**

39 Off-road vehicles (ORV) present many possibilities for accessing remote areas where there are no  
40 formed roads. However, the potentially adverse effects of these activities create challenges for  
41 environmental managers who must identify the existence, extent and severity of associated impacts,  
42 and design appropriate responses. Moreover, vehicular access can also facilitate other valued  
43 activities such as camping, hunting and fishing, leading to the need to consider the relative merits of  
44 vehicular access in various forms and its relationship with established objectives. This study  
45 investigates changes to vehicular beach access generated by the reconfiguration of a natural  
46 landscape. These changes were caused by the 7.8 M<sub>w</sub> Kaikōura earthquake that struck the east coast  
47 of New Zealand in November 2016, affecting over 130 km of coastline spanning two local  
48 government jurisdictions. The area included a c. 40 km section of the Marlborough region  
49 characterised by extensive sandy and mixed sand-gravel beaches that are the focus of this study.  
50 This relatively remote area is known for its wild and scenic values and has few road access points  
51 (Figure 1).

52 The earthquake caused a series of complex ruptures and fault movements associated with a  
53 highly variable pattern of vertical displacement (Hamling et al. 2017; Holden et al. 2017; Xu et al.  
54 2018). At the coastline, this displacement was mostly uplift by as much as 6 m (Clark et al. 2017;  
55 Orchard et al. 2021), leading to the widening of beaches. Soon after the earthquake, vehicle  
56 movements along the coast increased as an anthropogenic side-effect of topographical changes that  
57 improved access opportunities. These changes represent initially un-noticed and un-managed  
58 human responses with an apparent gain for recreational vehicle users, but with potentially  
59 undesirable aspects that remained unexplored. Moreover, there are transferable potential lessons  
60 for other large-scale landscape changes in disaster recovery and resource management contexts  
61 elsewhere.

62 In many parts of the world, ORV access is reportedly increasing and in many cases their  
63 cumulative effects have not been adequately addressed or managed. For example, Priskin (2003)  
64 reported that the number of ORV access points had increased by 115% between 1965 and 1998 on  
65 the Central Coast of Western Australia. Drivers may also drive along the edge of existing formed  
66 tracks, widening the area of disturbance (Davenport & Davenport 2006). Studies from Queensland

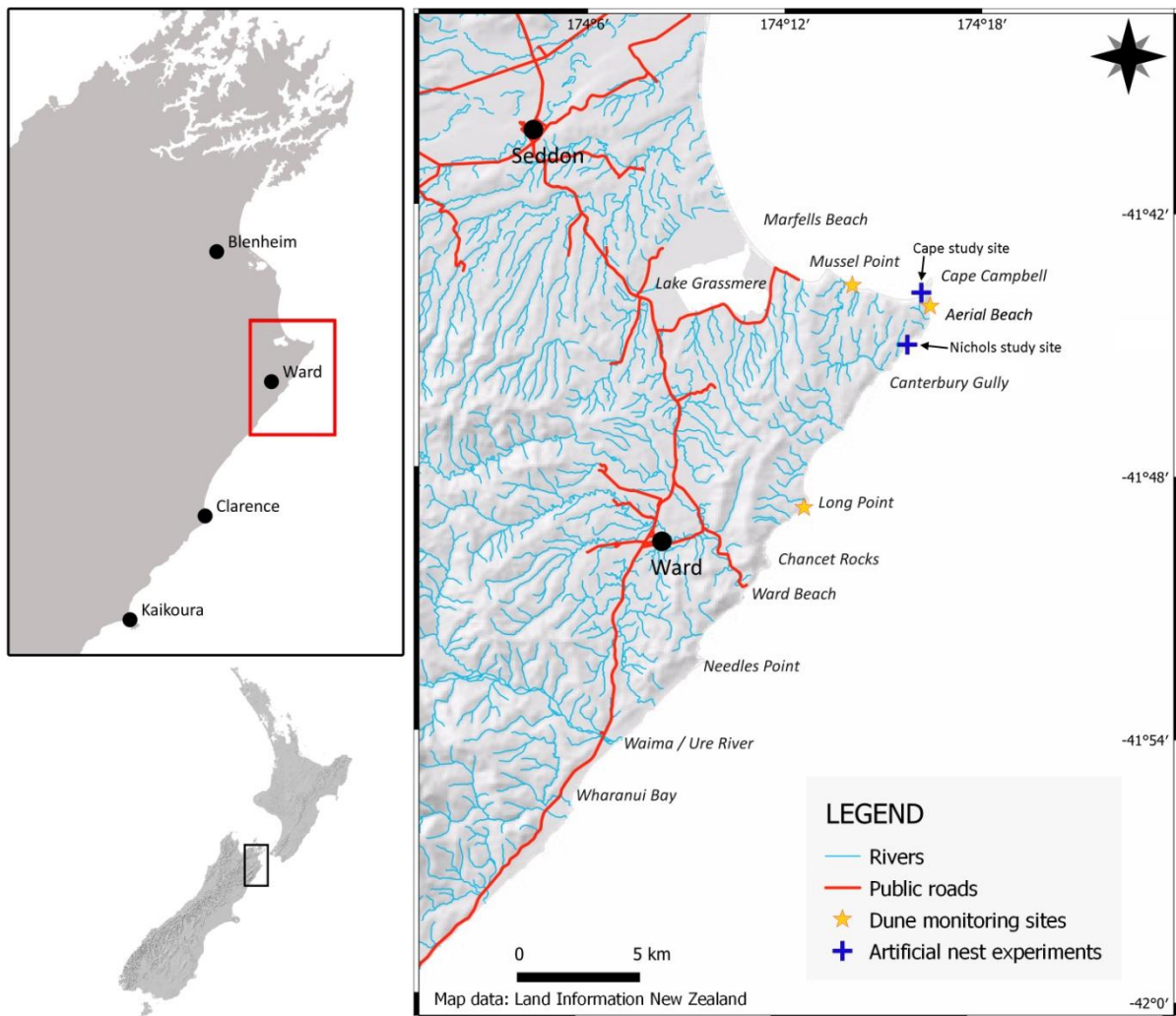
67 have reported more concentrated vehicle traffic on the upper beach, with the lower beach being  
68 used to varying degrees when exposed on lower tides (Schlacher & Thompson 2007).

69 The severity of ORV impacts depends on factors such as the frequency, timing and type of  
70 traffic, composition and vulnerability of biological communities, and the precise location of vehicle  
71 tracking in relation to sensitive areas. Several studies on ORVs have reported detrimental impacts on  
72 the structure and function of sandy beach ecosystems. For example, beaches with ORV traffic may  
73 have dunes set further back from the high tide line than beaches closed to ORVs (Houser et al.  
74 2013), indicating impacts on the formation of foredunes in areas they would otherwise occupy.  
75 Direct impacts of ORV traffic include sand compaction, erosion and vegetation loss leading to dune  
76 destabilisation (Davenport & Davenport 2006; Groom et al. 2007; Hosier & Eaton 1980; Houser et al.  
77 2013). Because dunes act as a barrier against storm surge and sea-level rise, damage to them can  
78 also reduce protection for housing and other infrastructure to landward (Orchard & Schiel 2021).  
79 The potential for damage to plant communities is an important consideration for ORV impact  
80 assessments because of the sand-trapping properties of dune plants (Martinez et al. 2016). The  
81 vegetation-mediated impacts of ORVs are potentially more important than direct erosion effects of  
82 vehicle tracking since much larger volumes of sand may be released from dune systems by wind  
83 erosion in the absence of plant cover. In New Zealand, where invasive marram grass (*Ammophila*  
84 *arenaria*) has displaced native sand-binders, ORV impacts may also contribute to the demise of  
85 native plant communities and their associated fauna, thereby creating an additional conservation  
86 issue (Stephenson 1999).

87 Shorebirds are among the most studied wildlife in relation to ORV impacts because of the  
88 potential vulnerability of key life stages such as nesting. Crushing by vehicles is particularly  
89 problematic for species that lay camouflaged eggs in cryptic nests such as 'scrapes' on beaches  
90 (Stephenson 1999; Weston et al. 2012). Disturbance by vehicles may also cause birds to reduce their  
91 feeding time or increase time away from the nest (Defeo et al. 2009). Studies in Queensland found  
92 that only 34% of ORV drivers slowed down or changed course when they encountered a shorebird,  
93 and that partial beach closures were effective in reducing the rate of egg crushing (Weston et al.  
94 2012; Weston et al. 2014). Additionally, numerous studies have shown adverse ORV impacts on  
95 beach invertebrates, including crustaceans and shellfish that are vulnerable to crushing (Davies et al.  
96 2016; Lucrezi & Schlacher 2010; Moss & McPhee 2006; Schlacher et al. 2007; Schlacher et al. 2008a).

97 Methods to control ORV impacts include the use of non-regulatory signage in sensitive areas  
98 (Supplementary Material Figure S1). Additionally, statutory measures may be imposed through legal  
99 instruments such as bylaws. These have been implemented for the control of vehicle impacts on  
100 several beaches in New Zealand to date, although are often hotly contested by stakeholders. In this

101 case study, post-disaster responses for coastal management have included the development of a  
102 proposed beach vehicle bylaw (Marlborough District Council 2021), and related aspects will require  
103 integration within longer-term planning arrangements. To support these initiatives our research  
104 programme included a suite of social-ecological investigations across a spatially extensive coastal  
105 area. Our objectives included the quantification of beach profile changes, vehicle tracking patterns,  
106 dune system responses and distribution of nesting grounds for banded dotterel (*Charadrius bicinctus*  
107 *bicinctus*) as an indicator species currently listed as 'nationally vulnerable' in the New Zealand Threat  
108 Classification System (Robertson et al. 2017).  
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111

112 **Figure 1** Overview of the study area on the east coast of the South Island of New Zealand.

## 113 **2. Methods**

### 114 **2.1 Shoreline characteristics and change**

115 The study area is a contiguous 36 km section of the Marlborough coastline stretching from the  
116 Waima / Ure River in the south to Marfells Beach in the north (Figure 1). This area marks the  
117 northern extent of coastal uplift associated with the Kaikōura earthquake. The degree of uplift in the  
118 study area was in the 1-3 m range and mainly associated with movements on the Needles Fault  
119 (Clark et al. 2017; Hamling et al. 2017; Holden et al. 2017). The coastal environment in this area  
120 differs considerably from the predominantly rocky Kaikōura coast further south and includes  
121 extensive sandy beaches and dune systems interspersed with headlands. Coastal reef substrates  
122 include relatively soft sedimentary rock platforms than have experienced high rates of erosion  
123 (Schiel et al. 2019). The combination of unique geology and a generally dry climate are associated  
124 with high faunal and floristic diversity and the coast is an important area for migratory bird species  
125 (de Lange et al. 2013; Jones & Hutzler 2002; Marlborough District Council 2021).

126 Pre-quake shorelines for the study area were digitised for the spring high tide line and dune  
127 system toe on the high tide beach using visible markers such as vegetation limits, strand lines and  
128 drying heights visible in high resolution aerial photography acquired 2015. This imagery was  
129 comparable to immediate post-quake data acquired two years afterwards, with the pixel size of both  
130 datasets being 0.2 m (Table 1). Assessment of the pre-quake landscape was also assisted by  
131 comparison with a 2002 aerial imagery dataset that happened to be captured at high tide, creating a  
132 useful reference for inundation levels at many points along the coast (Table 1). Additionally, the  
133 whole area was surveyed on foot in 2018 and 2019 and GPS coordinates captured for a wide range  
134 of landmarks and reference points including pre-quake vegetation zonation patterns and the  
135 position of new recruitment zones. Contour intervals at 0.1 m were extracted from the post-quake  
136 LiDAR and the 0.6 m contour was adopted as a representative position for the estimated high tide  
137 line based on the abovementioned visual clues. This vertical elevation is equivalent to a mean high  
138 water spring tidal height of ~0.4m NZVD and an additional 0.2 m (vertical) swash zone (Land  
139 Information New Zealand 2021).

140 **Table 1** Aerial imagery and LiDAR datasets

141  
142

(a) Aerial imagery

Dataset	Acquisition date	Supplier	Commissioning agency	Imagery specifications	
				Resolution	Format
Marlborough 1m Rural Aerial Photos (2002)	Various dates in 2002	Terralink International	Marlborough District Council	100 cm pixel	Black and White GeoTIFF
Marlborough 0.2m Rural Aerial Photos (2016)	7 January 2016 - 7 March 2016	AAM NZ Ltd	Marlborough District Council	20 cm pixel	3-band (RGB) GeoTIFF
Post-quake Aerial Imagery <sup>†</sup>	3 December 2016 - 6 January 2017	AAM NZ Ltd	Land Information New Zealand	20 cm pixel	3-band (RGB) GeoTIFF

143

144 (b) LiDAR data

Dataset	Acquisition date	Supplier	Commissioning agency	Accuracy specification (m)	
				vertical	horizontal
Post-quake Airborne LiDAR <sup>†</sup>	3 December 2016 - 6 January 2017	AAM NZ Ltd	Land Information New Zealand	±0.10	±0.50

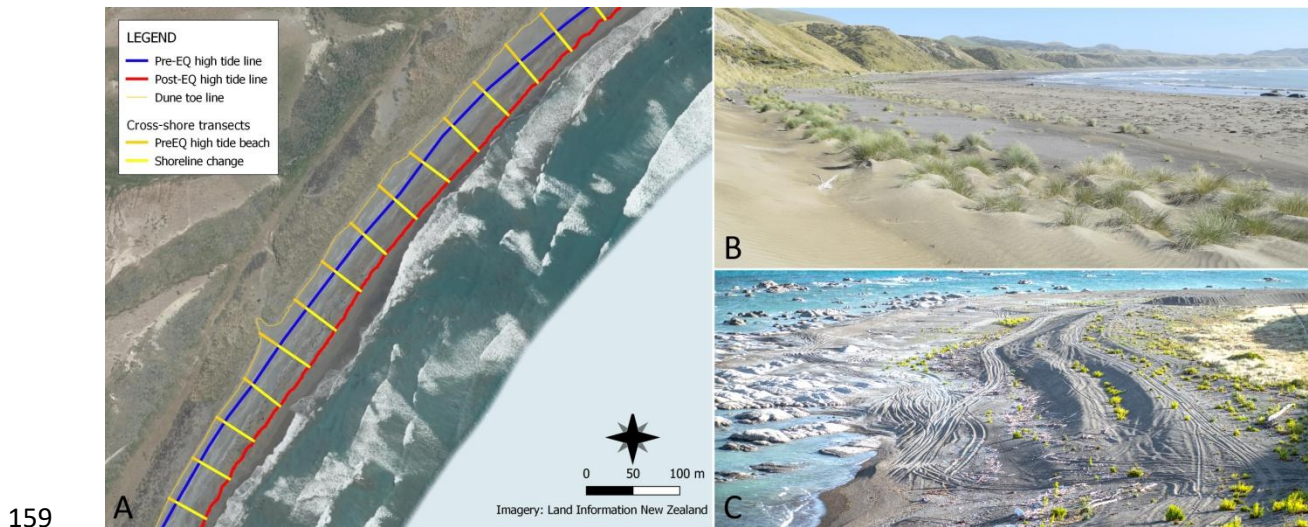
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146 <sup>†</sup> captured concurrently

147

148 A set of 720 cross-shore (perpendicular to the shoreline) transects were generated at 50 m intervals  
 149 along a smoothed baseline prepared from the pre-quake high tide line using the *ambur* package in R  
 150 (Jackson et al. 2012). Intersection analyses in a GIS environment were used to compute cross-shore  
 151 beach width and shoreline changes in relation to the timing of the earthquake (Figure 2). To examine  
 152 spatial co-occurrence patterns, a complementary set of belt transects, each 50 m wide at the  
 153 baseline, was generated in a northward direction from the first transect origin at Waima River in the  
 154 south. Substrate types for the belt transects were classified using field data according to four classes  
 155 (reef, boulder, mixed sand-gravel, sand) that represent the predominant substrate on the high tide  
 156 beach. Collectively, these belt transects covered the entire coast and were of a sufficient cross-shore  
 157 dimension to intersect with all other field survey and remote-sensed data as described in the  
 158 sections below.





159

160 **Figure 2** (A) Example of the shoreline change sampling set-up showing cross-shore transects at 50 m spacing  
161 and two example calculations overlaid on post-quake aerial imagery. The area in (A) is shown in (B) located  
162 north of Long Point where the beach has widened considerably and new dunes have formed. (C) Post-quake  
163 vehicle tracking on the high tide beaches between Cape Campbell and Canterbury Gully.

164

## 165 2.2 Off-road vehicle tracking

166 Off-road vehicle (ORV) tracking measurements were made at periodic intervals throughout the study  
167 area and included whole-coast surveys completed in the summers of 2018 and 2019. The width of  
168 visible vehicle tracks was measured in the cross-shore direction to the nearest metre. Monitoring  
169 points included at the location of changes in vehicle tracking patterns and at the position of all  
170 shorebird nesting territories (described below). This measurement reflects the distance between the  
171 tyre tracks of individual vehicles or, in the case of overlapping tracks, the dimension between the  
172 outer tyre marks, summed across the shore profile. Only tracking above the high tide line was  
173 considered, to avoid biases introduced by observations of tracking below the high tide line which are  
174 influenced by survey timing in relation to the tide. Additional information collected included  
175 evidence of preferred routes, such as where tracks were seen to converge or fan out in response to  
176 barriers and topographic changes, and the location of access points and turnaround areas.

177 **2.3 Shorebird nesting sites**

178 *Census surveys*

179 Surveys of banded dotterel nesting territories were completed for the entire coastline in November  
180 2018 and 2019. Coordinates were recorded using handheld GPS receivers for the estimated  
181 midpoint of nesting territories based on bird behaviour. Visual clues used to identify territories  
182 included characteristic ward-like behaviour, which includes the 'escorting' of intruders to beyond the  
183 territorial range, in addition to distraction behaviour such as broken wing displays that are indicators  
184 of a nest or chicks nearby. In areas supporting several nesting territories the boundaries were  
185 discriminated using two observers following the movements of adjacent breeding pairs. Additional  
186 searches were made for nests at two locations (Ward Beach and Canterbury Gully) to assess the  
187 cross-shore location of nests on the high tide beach with a focus on establishing whether newly  
188 created habitat associated with beach widening was being utilised for nesting sites. Nesting territory  
189 mid-points were overlaid on the belt transects to assess relationships with substrate types and  
190 vehicle tracking density.

191

192 *Artificial nest experiment*

193 To provide a more direct test of the risk of vehicle strikes, an artificial nest experiment was  
194 conducted at two sites after the breeding season. These experiments used quail eggs which are very  
195 similar to banded dotterel eggs in size and colouration and were deployed in shallow scrapes to  
196 provide a close representation of banded dotterel nests (Supplementary Material, Figure S2). The  
197 Cape Campbell site represented a location with a high traffic density of c.70% tracking on a relatively  
198 narrow high tide beach 20-25 m wide. The Nicholls site represented a location with a moderate  
199 traffic density of c.40% tracking on a high tide beach 60 m wide. At each site, six representative  
200 territories were established, each occupying 50 m in the long-shore direction. To examine the  
201 potential consequence of nesting locations in old vehicle tracks (which is relative common for  
202 banded dotterel), potential nest sites within each territory were stratified according to their  
203 previous tracking status (tracked / untracked). Four nests were placed at random locations within  
204 each territory using a 1 m grid and applying a minimum distance to any vehicle track of 2 m to define  
205 the untracked class. Additionally, a buffer of 5 m minimum was maintained from the upper and  
206 lower limits of the high tide beach to reduce confounding edge effects. Both study sites were mixed  
207 sand-gravel beaches and the area around each nest was smoothed to facilitate the detection of  
208 predator tracks.

209 All nests were monitored weekly for nest failures which were classified according to five  
210 classes of threats (vehicle strike, trampling by horses, avian predation, small mammal predation, and



211 unknown / other predation) based on visual clues. Vehicle strikes were readily identifiable from  
212 recent tracking and the observation of crushed nests, as was the single example of trampling by  
213 horses. Avian predation was evidenced by the observation of peck marks with the egg shell being  
214 typically crushed inwards (Supplementary Material, Figure S2). Small mammal predation was  
215 recorded at a few nests where it was associated with partially eaten eggs. Other predation was the  
216 class assigned to the disappearance of eggs, often accompanied by mammalian tracks being  
217 observed in the vicinity of the nest. These losses are likely attributable to larger and more vigorous  
218 predators such as mustelids or feral cats. As the data followed a non-normal distribution, Kruskal  
219 Wallis tests were used to assess the effects of territory and previous tracking status on nest losses at  
220 each study site. To incorporate the effect of site within the analysis and improve statistical power  
221 using the combined dataset we fitted Generalized Linear Mixed Models with binomial error  
222 distributions to both total nest losses and nest losses by vehicle strikes. In both GLMMs, we treated  
223 time (week) as a random effect and previous tracking status, site, and their interaction as fixed  
224 effects. Analyses were produced in R version 4.0.3 (R Core Team 2021), using the *tidyverse* package  
225 (Wickham et al. 2019), and *glmer* function in the *lme4* package for the GLMMs (Bates et al. 2008).

226

#### 227 **2.4 Dune system responses**

228 Field transects were established at three sandy beaches located east of Mussel Point, south of Cape  
229 Campbell (Aerial Beach) and south of Long Point (Figure 1). Each of these beaches supports the  
230 native sand-binder spinifex (*Spinifex sericeus*) and the post-quake response of spinifex dunes is of  
231 considerable interest for beach conservation objectives. At each monitoring site, two cross-shore  
232 transects were established from an origin point in the back dune swale. Each transect spanned the  
233 pre-quake foredune and newly developing dune system on the uplifted beach. The Mussel Point and  
234 Long Point sites featured spinifex remnants in the old dune system that were expected to colonise  
235 the new accommodation space created by uplift through the vegetative growth of runners from  
236 established plants. At the Aerial Beach site there were no spinifex remnants in the old dune system  
237 due to extensive invasion by marram grass along this section of coast, but several relatively large  
238 spinifex patches had become established on the high tide beach (likely reflecting recruitment soon  
239 after the earthquake). Measurements taken in the summers of 2018 and 2019 included the canopy  
240 cover and maximum height of vegetation within contiguous 2 x 2 m plots along the transect line that  
241 together provided a complete description of the vegetation composition within a 2 m belt across the  
242 dune profile. Canopy cover estimation followed the method of Hurst & Allen (2007) which considers  
243 the area within the perimeter of the canopy of each plant as seen in plan-view for each species of  
244 interest. Bare ground cover was recorded where it occurred and vehicle tracking measured as

245 detailed above. Ground elevation was measured at 1 m intervals along each transect using a  
246 combination of laser level and real-time kinematic (RTK) GPS surveys with a Trimble R8 GNSS  
247 receiver. Geodetic benchmarks were included within all RTK-GPS surveys and referenced to the New  
248 Zealand Vertical Datum (NZVD) (Land Information New Zealand 2016), to achieve an estimated  
249 vertical accuracy of  $3.5\text{mm} \pm 0.4\text{ ppm RMS}$ .

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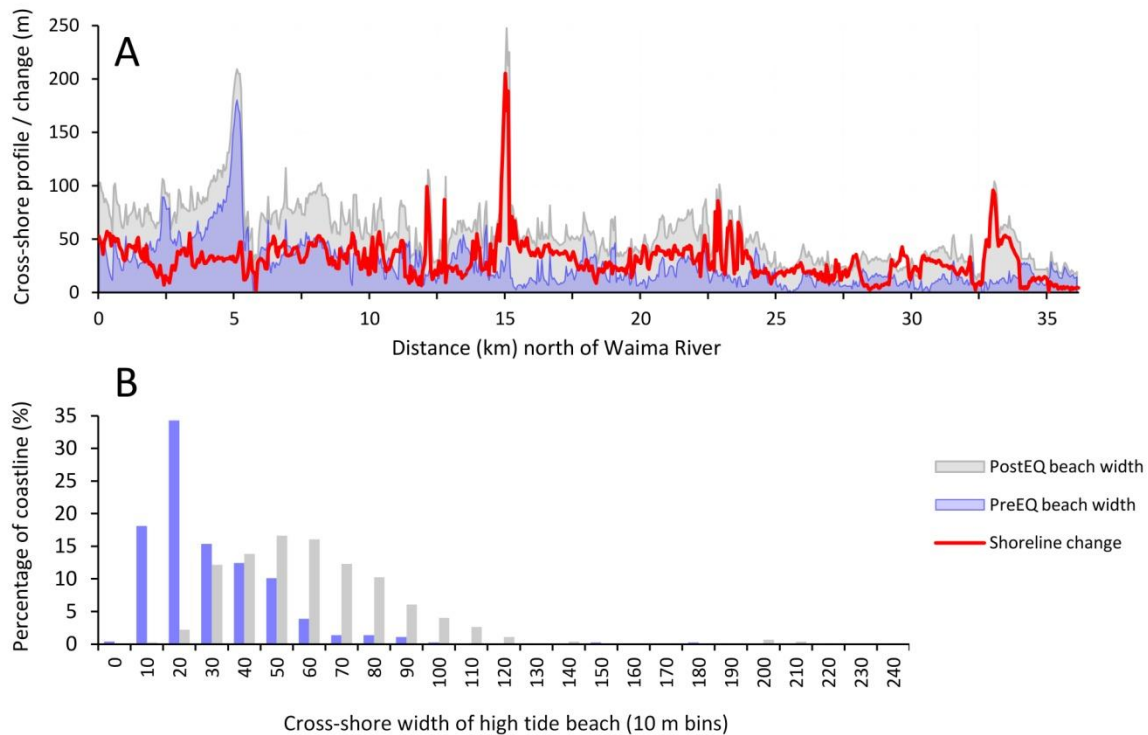
## 251 **3. Results**

### 252 **3.1 Shoreline change**

253 Seaward movement of the pre-quake high tide line was observed on all transects although the  
254 magnitude of the changes was highly variable, as expected, due to variable uplift (Figure 3A).  
255 Shoreline change ranged from 1.2 m - 205 m with a mean of  $30.6 \pm 0.7\text{ m (SE)}$ . Prior to the  
256 earthquake 90% of the coastline featured high tide beaches that were <50 m wide. This percentage  
257 was halved to 45% as a result of beach widening induced by coastal uplift. The mean cross-shore  
258 width of high tide beaches increased from  $25.6 \pm 0.9\text{ m}$  to  $56.2 \pm 1.2\text{ m}$ . Across all transects, the  
259 minimum width increased from zero to 7 m, and maximum width increased from 180 m to 247 m  
260 (Figure 3B).

261 Changes to high tide beaches were considerable. Prior to the earthquake, some sections of  
262 the coast had no high tide beach ( $n=21$  transects), while 50 transects (2.5 km of coastline) were <5 m  
263 wide, and 157 transects (7.8 km of coastline) were <10 m wide. These narrow sections of coast  
264 included rocky headlands at Mussel Point, Cape Campbell, Chancet Rocks and Needles Point (Figure  
265 1). Following the earthquake, there were no transects with high tide beaches <5 m wide and only  
266 two <10 m wide, both located at Mussel Point. These results indicate that pre-quake locations with  
267 topographical barriers to ORV traffic at high tide had been widened sufficiently to facilitate vehicle  
268 access, given that at least 10 m of high tide beach had become available nearly everywhere. These  
269 changes are attributable almost solely to uplift of the coastline since the LiDAR data and imagery  
270 used to assess post-quake conditions was acquired within a few days of the earthquake.

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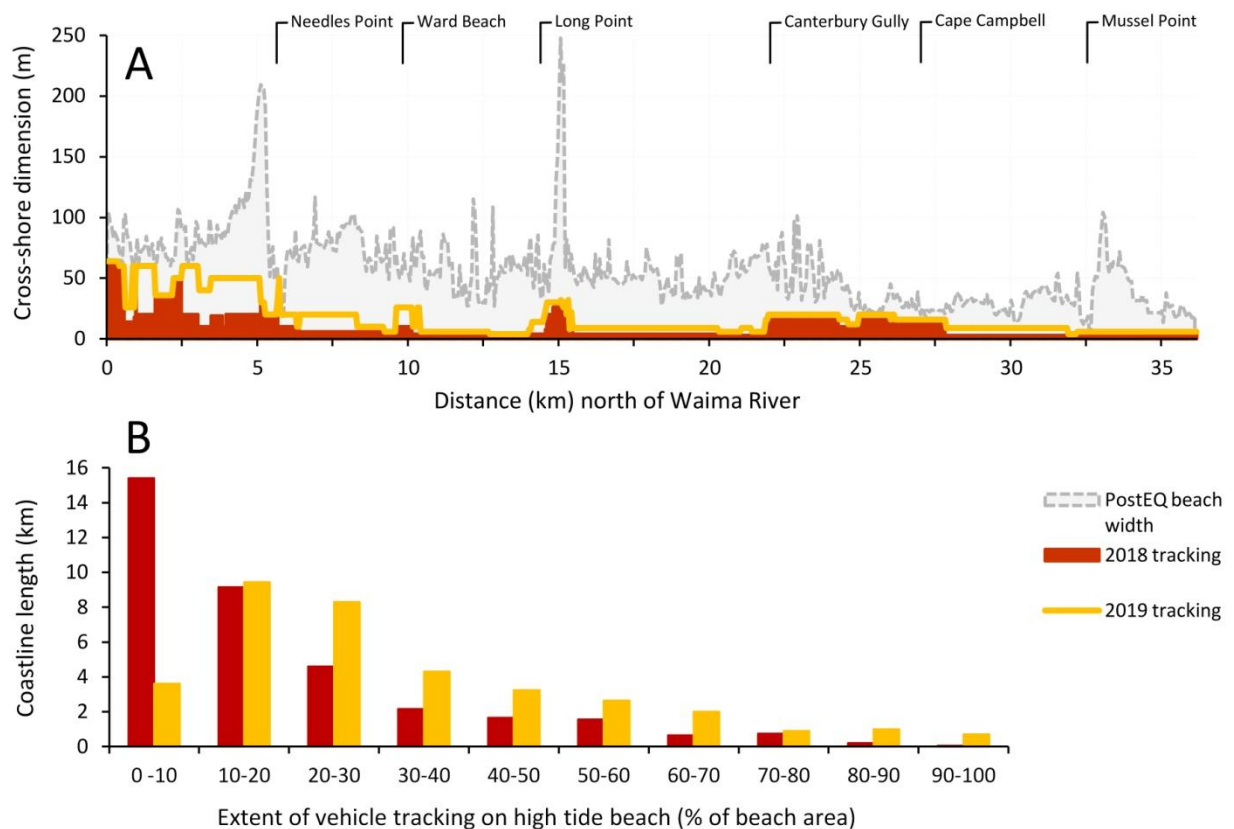
273 **Figure 3** (A) Cross-shore width of the high tide beach before and after the 2016 Kaikōura earthquake along 36  
274 km of coastline from the Waima River in the south to Marfells Beach in the north. (B) Histograms of the high  
275 tide beach width calculated for 10 m beach-width increments. The X-axis labels represent the upper value of  
276 each 10 m bin. An additional 'zero width' bin is also included.

### 277 3.2 Vehicle tracking

278 Vehicle tracking patterns have changed considerably along the coast, although to varying degrees.  
279 These are most pronounced at sections of the coastline characterised by steep rocky substrates on  
280 the cross-shore profile and steep hill country above that previously created a barrier to most forms  
281 of ORV traffic. Changes in the north of the study area (near Marfells Beach) had the greatest  
282 influence (Figure 4A). They involved only short sections of rocky coast at Mussel Point and headlands  
283 further east that experienced only modest uplift (c. 1m), but it was sufficient to expose a high tide  
284 ledges that facilitated all-tide access to Cape Campbell under most combinations of tide and swell.  
285 Accompanying changes in ORV usage have included relatively large volumes of traffic (e.g., daily  
286 ORVs >30 at Cape Campbell), becoming commonplace in previously remote areas that were seldom  
287 visited before.

288 The vehicle tracking pattern in 2018 showed extensive tracking in the south of the study  
289 area between Waima River and Needles Point, but this had also been the case before the  
290 earthquake due to the river mouth being a popular vehicle access point (Figure 4A). To the north of  
291 Needles Point, tracking was reduced to 6-10 m which is generated mostly by vehicles travelling south  
292 from Ward Beach, a popular public access point located 4km further north. Similarly, Chancet Rocks,

293 located 2 km north of Ward Beach, prevented most ORVs from travelling further north from the  
294 Ward Beach access point. The stretch of coast immediately north of Chancet Rocks featured the  
295 lowest level of vehicle tracking in the study area in both years (equivalent to 1 -2 vehicle tracks).  
296 However, many of those vehicle movements originated from Marfells Beach located 25 km to north,  
297 in addition to a few private land access points. The cross-shore tracking pattern partly reflects  
298 popular route choices to avoid obstacles, such as at Long Point where reef platforms are present  
299 lower in the intertidal zone. Between 2018 and 2019 the whole-coast average increased 73% from  
300  $10.0 \pm 0.4$  m to  $17.3 \pm 0.6$  m of tracking on the cross-shore profile, despite the minimum and  
301 maximum values staying unchanged at 3 m and 64 m respectively. The associated change in tracking  
302 density (taking into account the width of the beach) was a 67% increase from  $19.5 \pm 0.4\%$  to  $32.6 \pm$   
303  $0.8\%$  of the high tide beach area (Figure 4B).  
304



305

306 **Figure 4** Vehicle tracking patterns across 36 km of coastline between Waima River and Marfells Beach  
307 measured over two consecutive summers following the November 2016 Kaikōura earthquake. (A) Cross-shore  
308 high tide beach width and extent of vehicle tracking. (B) Histogram of vehicle tracking extent showing the  
309 percentage of beach and length of coastline affected.

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311 **3.3 Banded dotterel nesting sites**

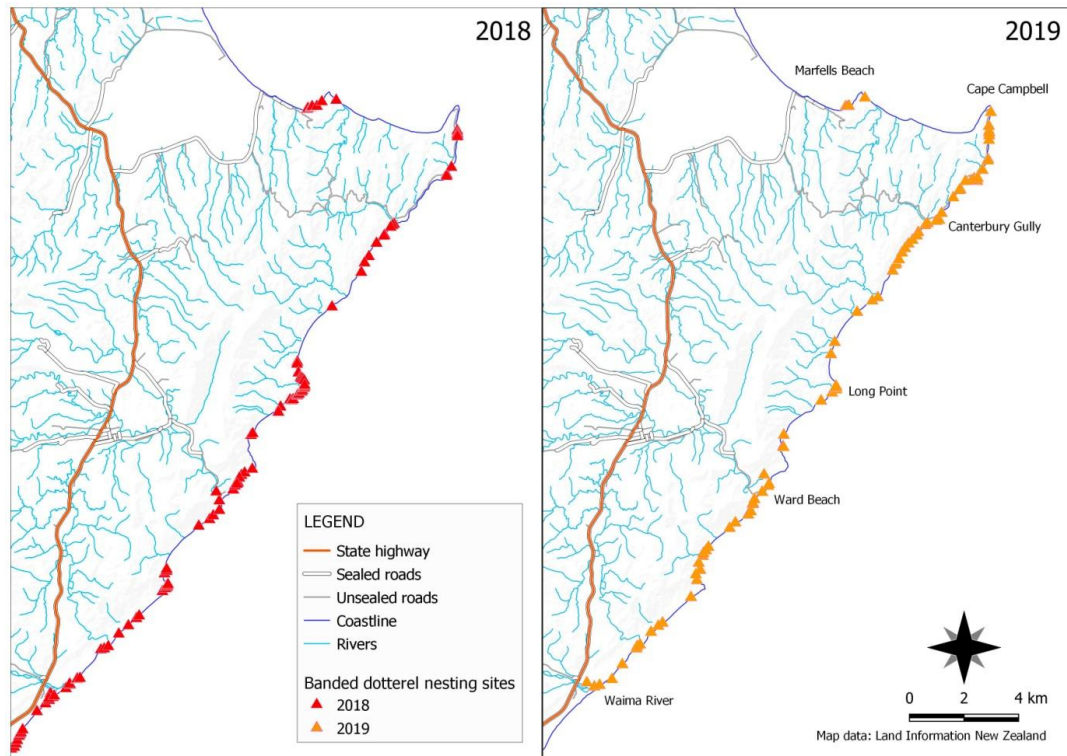
312 *Census survey*

313 A total of 60 nesting territories were identified in the 2018 survey, and 69 territories in 2019 (Figure  
314 5). Although the spatial distribution varied slightly, locations with high nest density were very similar  
315 between years. Some of the changes observed were a higher number of nesting pairs south of Cape  
316 Campbell on both sides of Canterbury Gully, and lower numbers at Long Point in 2019 in comparison  
317 to 2018 (Figure 5). Beaches characterised by mixed sand-gravel substrates supported the majority of  
318 nesting territories in both years (73.3% and 76.8% for 2018 and 2019, respectively) (Figure 6A).  
319 Territories were also recorded on sandy beaches (21.7% and 17.4% of nesting territories for 2018  
320 and 2019, respectively), in locations such as the bay north of Long Point and at Mussel Point.  
321 However, the pattern of substrate fidelity is somewhat blurred by the heterogeneous nature of  
322 many beaches that feature patches of various substrate types. On the finer sand beaches, for  
323 example, dotterels are still likely to find areas of coarser gravels, which provide more suitable  
324 camouflage for the placement of nests.

325 Nesting sites may be located anywhere on the high tide beach, including close to the new  
326 post-quake high tide line. This indicates that beach widening has increased the area of suitable  
327 habitat. All dotterel nesting territories were exposed to some degree of disturbance from motor  
328 vehicles, as evidenced by tracking patterns (Figure 6B). The degree of spatial overlap is indicated by  
329 the tracking density within dotterel territories which varied by 3–90% in 2018, and 3–99% in 2019.  
330 The mean tracking density increased from 19.5 to 32.3% between years within dotterel territories. A  
331 histogram of these data shows a decrease in frequency in the 0-10% bin and a relatively consistent  
332 increase across other bins, indicative of a gradual increase in the density of tracks in all of the  
333 dotterel nesting areas (Figure 6B). It is important to note that these anecdotal tracking patterns  
334 record only previous vehicle movements and may include old tracks that persisted from the previous  
335 nesting season. Field observations of vehicle movements made on various dates and times suggest  
336 that the actual volume of traffic was similar between years.

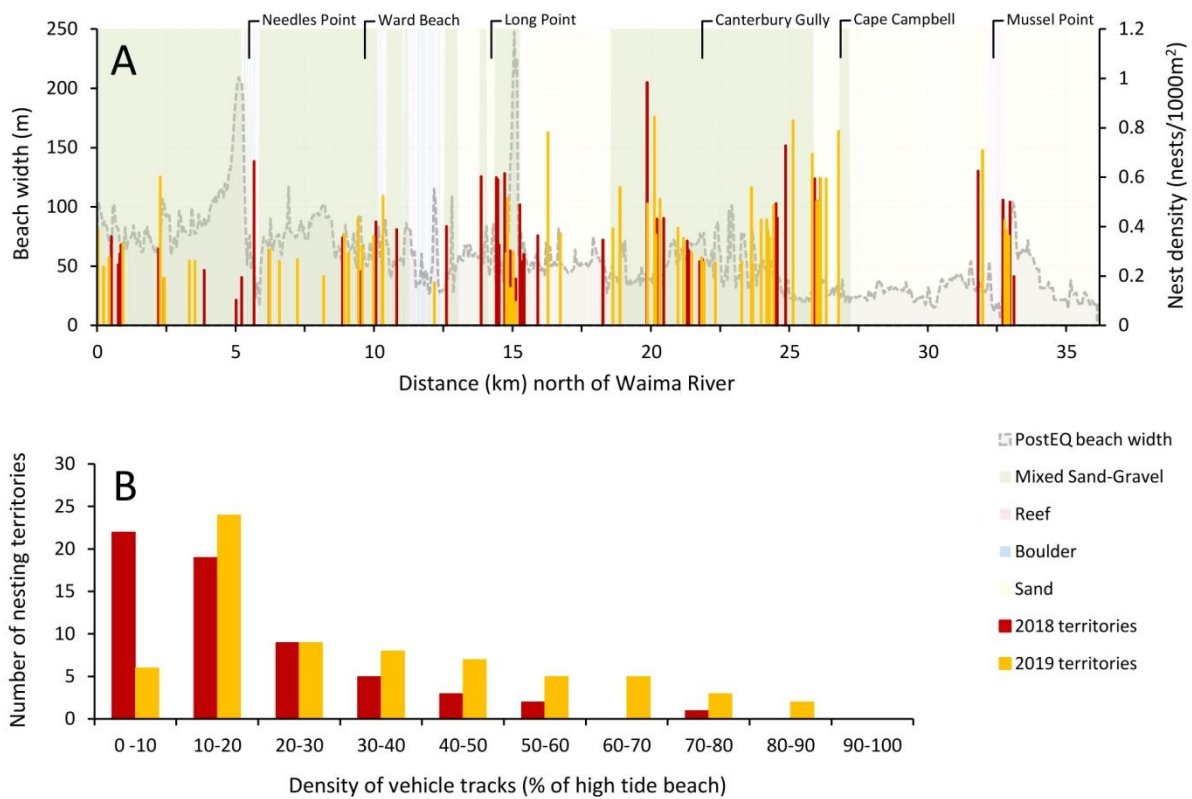
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339 **Figure 5** Distribution of banded dotterel (*Charadrius bicinctus bicinctus*) nesting territories across 36 km of  
 340 coastline between Waima River and Marfells Beach in 2018 and 2019.



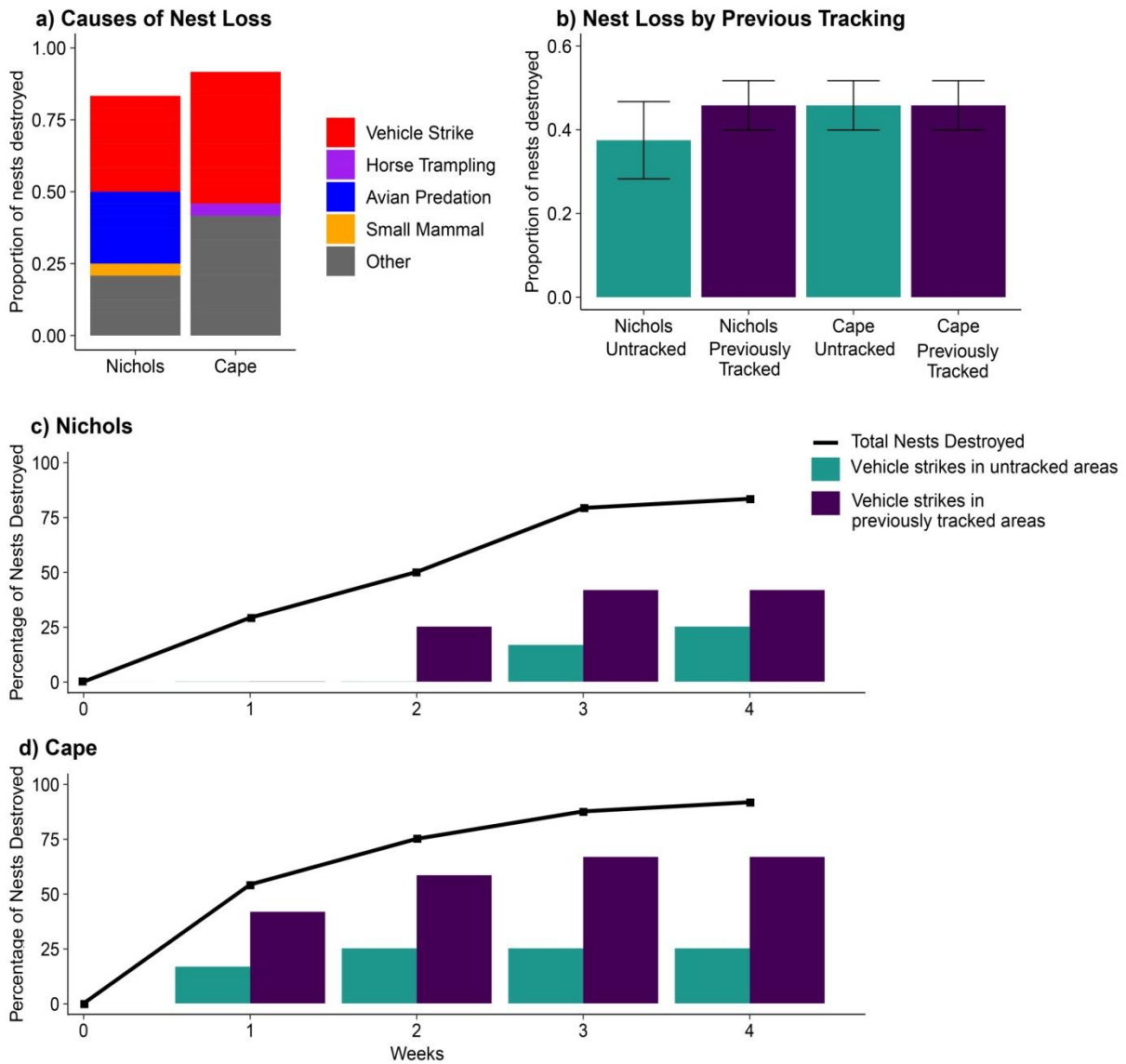
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342 **Figure 6** (A) Density of banded dotterel (*Charadrius bicinctus bicinctus*) nesting territories across 36 km of  
 343 coastline between Waima River and Marfells Beach overlaid on the high tide beach and dominant substrate  
 344 type. (B) Density of vehicle tracking in nesting territories expressed as percentage of the high tide beach.

345 *Artificial nest experiment*

346 Over the typical incubation period of four weeks there was a high nest loss rate at both study sites,  
347 with 92% of nests failing at the Cape site, and 83% at Nichols (Figure 7). Generalized Linear Mixed  
348 Models of total nest losses over time showed that there was a significant difference between the  
349 study sites ( $Z = 2.897$ ,  $p = 0.004$ ), with the higher tracking density site (Cape) showing a higher nest  
350 loss rate (Table 2). This finding is consistent with a correlation between actual vehicle movements  
351 during the study period and the previous density of vehicle tracking that was used to characterise  
352 and select the study sites. The Cape site also had a higher overall number of vehicle strikes and they  
353 occurred earlier in the study period. Vehicle crushing was the single biggest causal factor, accounting  
354 for 50% of the nest losses at the Cape site and 40% at Nichols. The GLMMs, however, showed no  
355 significant difference between sites for nest losses by vehicle strikes ( $Z = 1.634$ ,  $p = 0.102$ ). This  
356 partly reflects the smaller difference between sites for the vehicle strike rate versus overall losses,  
357 and variance introduced by the previously tracked and untracked nest locations (Figure 7). Previous  
358 tracking status was found to be a significant factor for vehicle strikes as the cause of failure ( $Z =$   
359  $3.495$ ,  $p < 0.001$ ), but not for total nest losses ( $Z = 1.047$ ,  $p = 0.295$ ). This is consistent with  
360 preferential use of existing vehicle tracks by ORVs as expected. Nonetheless, the relatively high nest  
361 loss rates at locations that were previously untracked shows that vehicles are continuing to explore  
362 new areas. At the Nichols site, 38% of all nests were lost to vehicles on previously untracked parts of  
363 the beach, and the situation was worse (46% of nests) with higher traffic density at the Cape study  
364 site.

365



366

367 **Figure 7** Artificial nest experiments at two study sites on the Marlborough coast where beaches have recently  
 368 widened due to uplift from the 2016 Kaikōura earthquake. Four nests were located within each of six  
 369 territories (n=24 nests per study site) and stratified by the presence of previous vehicle tracking (tracked /  
 370 untracked) at the nest location. The Cape site is located near Cape Campbell where the high tide beaches are  
 371 narrow. The Nichols site features wider high tide beaches and is located 4 km to the south (see Figure 1).

372 **Table 2** Statistical comparisons for artificial nest experiments.

373  
374

(a) Kruskal-Wallis tests on nest losses after 4 weeks.

Independent variable	Dependent variable	$\chi^2$	df	p†
Territory	Total nest losses - all sites	17.889	11	0.0841
Territory	Total nest losses - Nichols	5.75	5	0.3313
Territory	Total nest losses - Cape	10.455	5	0.06333
Previous tracking status	Total nest losses - all sites	0.8519	1	0.356
Previous tracking status	Total nest losses - Nichols	1.15	1	0.2835
Previous tracking status	Total nest losses - Cape	0	1	1.000
Previous tracking status	Nest losses from vehicle strikes - all sites	4.1797	1	<b>0.0409*</b>
Previous tracking status	Nest losses from vehicle strikes - Nichols	0.7188	1	0.3966
Previous tracking status	Nest losses from vehicle strikes - Cape	4.021	1	<b>0.0449*</b>

375  
376  
377

(b) Generalized Linear Mixed Model of total nest losses over time (random effect) with fixed effects of 'previous tracking status' and 'site'.

	Estimate	SE	Z	p†
Intercept	1.1913	0.6039	1.973	<b>0.0485*</b>
Site	1.4067	0.4855	2.897	<b>0.0038**</b>
Previous tracking status	0.5538	0.5289	1.047	0.2950
Site x Previous tracking status	0.9806	0.7184	1.365	0.1722

378  
379  
380

(c) Generalized Linear Mixed Model of nest losses from vehicle strikes over time (random effect) with fixed effects of 'previous tracking status' and 'site'.

	Estimate	SE	Z	p†
Intercept	1.2747	0.4273	2.983	<b>0.0029***</b>
Site	0.9641	0.5901	1.634	0.1023
Previous tracking status	1.6272	0.4655	3.495	<b>0.0005***</b>
Site X Previous tracking status	0.4317	0.7412	0.582	0.5602

381 † significant results at  $\alpha = 0.05$  shown in **bold**. \* =  $p < 0.05$ , \*\* =  $p < 0.01$ , \*\*\* =  $p < 0.001$

382

### 383 3.4 Dune responses

384 New dunes were observed along many stretches of coastline throughout the study area. Despite an  
385 abundance of new accommodation space, however, the speed and extent of new dune formation  
386 varied due to differences in the rate of vegetation establishment and interactions with sand supply.  
387 At the beach monitoring sites the width of the new dune zone varied from 10.5 m at Mussel Point to  
388 over 30 m at Aerial Beach and Long Point, and increased only slightly between the 2018 and 2019  
389 summers (Table 3). The widest new dune zone (38 m) was recorded at Aerial Beach in 2019. This site  
390 has a relatively flat cross-shore profile and exemplified the general pattern of new dune  
391 development elsewhere involving sand deposition above the post-quake high tide line, often

392 facilitated by vegetation establishment in the same areas. Two years after the earthquake, native  
 393 sand-binders including spinifex (*Spinifex sericeus*) and knobby club rush (*Ficinia nodosa*) had  
 394 established on many uplifted beaches along with invasive marram (*Ammophila arenaria*) and  
 395 seasonal flushes of sea rocket (*Cakile edentula*).

396 Vegetation height and percentage cover increased at all monitoring sites but to a variable  
 397 extent (Table 3). The largest increase of around 25% cover in new dune area was at Aerial Beach.  
 398 The narrower Mussel Point site had the highest percentage cover (29%) associated with the  
 399 establishment of a new belt of sand-binding vegetation above the old spring high tide line in front of  
 400 the old dune toe (Figure 8A,B). In contrast, the Long Beach site had only sparse cover in the new  
 401 dune zone (12-13%) that is characterised by a more sloping beach with mixed sand-gravel  
 402 substrates, which provides a harsh environment for plant establishment. New dune formation at the  
 403 site is partly dependent on vegetative growth from spinifex runners into the new space (Figure 8C-  
 404 E). This vegetative growth is creating new sand and debris accumulations that will likely assist wind-  
 405 blown recruits become established. However, the overall rate of vegetation colonisation appears to  
 406 be slower than the on the finer-grain beaches at Aerial Beach and Mussel Point.

407

408 **Table 3** Dune monitoring results from cross-shore transects at three beaches uplifted by the 2016 Kaikōura  
 409 earthquake.

Study sites	Cross-shore width of new dune <sup>†</sup> (m)	New dune height <sup>‡</sup> (m) above MHWS <sup>†</sup> (mean ± SE)	Percentage vegetation cover <sup>†</sup> (mean ± SE)	Maximum vegetation height <sup>†</sup> (m)	Vehicle tracking density in new dune zone <sup>†</sup> (%)
<i>Mussel Point</i>					
2018	10.5	1.97 ± 0.19	24.08 ± 8.98	0.56	11.1
2019	13	2.13 ± 0.16	28.29 ± 7.87	0.65	15.5
<i>Aerial Beach</i>					
2018	30.5	1.94 ± 0.16	9.00 ± 3.64	0.70	16.3
2019	38	1.95 ± 0.15	23.38 ± 4.73	0.75	30.3
<i>Long Point</i>					
2018	31	2.15 ± 0.30	11.74 ± 3.87	0.54	32.6
2019	31	2.16 ± 0.29	12.79 ± 3.63	0.65	48.5

410

411 <sup>†</sup> average value from n=2 transects at each site

412 <sup>‡</sup> measured at 1 m increments on transect line, MHWS = Mean High Water Springs

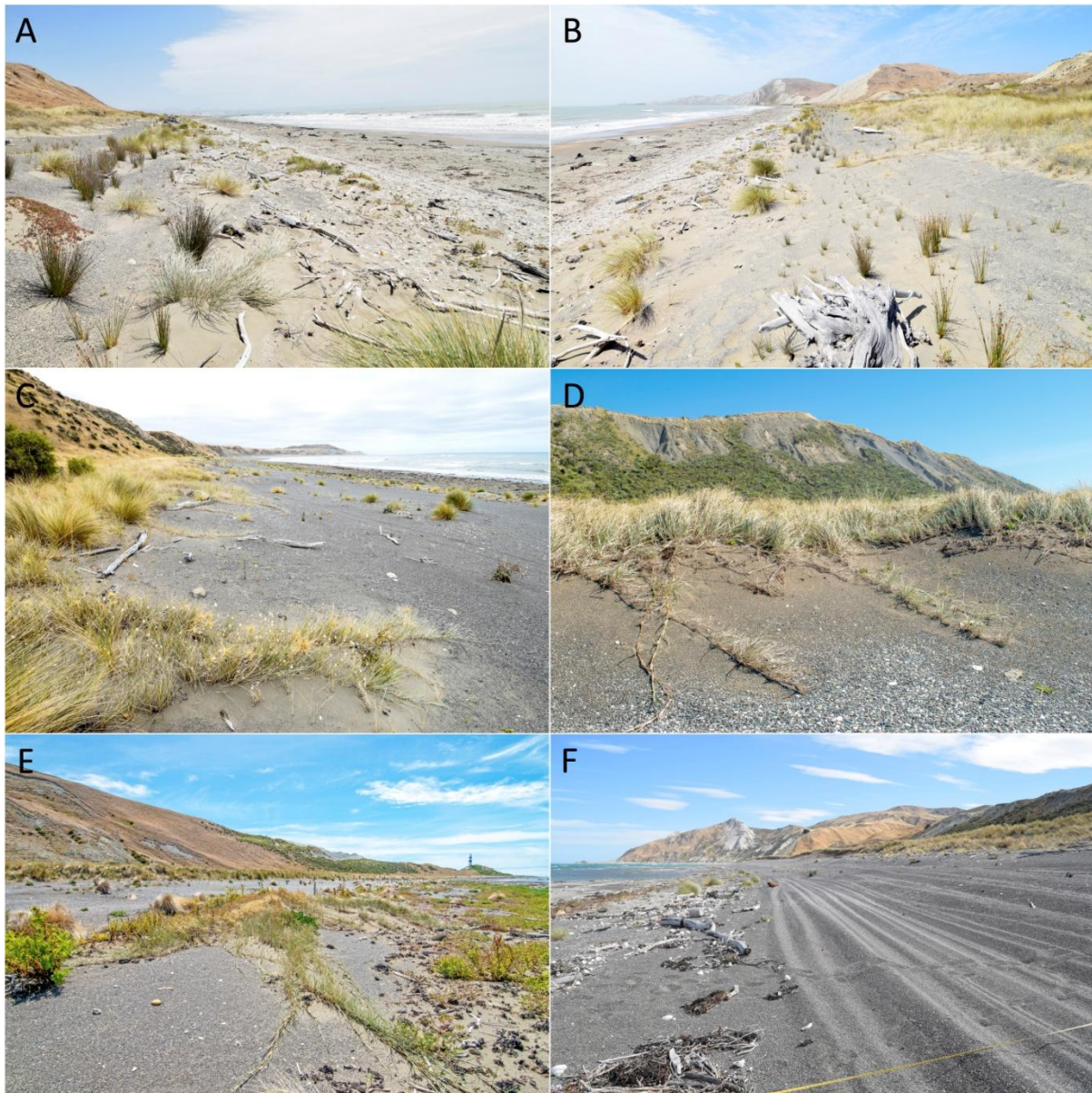
413

414 Although some sand accumulation had already occurred in the new dune zone in the two years since  
 415 the earthquake, further accumulation (as indicated by the average height of the profile), was only  
 416 modest at all sites between 2018 and 2019 (Table 3). The observed vegetation cover and height  
 417 increases did not appear to be sufficient to trap an appreciable quantity of additional sand between



418 years. This may reflect nuances in the relationship between sand supply and the sediment-trapping  
419 properties of vegetation, or reflect erosion effects and other interactions.

420         The relatively slow rates of vegetation recovery and new dune formation since the  
421 earthquake are influenced considerably by people. Vehicle tracking densities, which provide a spatial  
422 measure of the extent of disturbance, were as high as 50% at Long Point by 2019 (Figure 8F), and  
423 increased at all sites between 2018 and 2019 (Table 3). The lowest tracking density (15%) at Mussel  
424 Point also coincided with the highest vegetation cover, suggesting a link between recent vehicle  
425 disturbance and the establishment of new plants. The tracking patterns were oriented mainly  
426 parallel to the shoreline, indicative of ORVs being used for long-shore access, but there were also  
427 examples of tracks on relatively steep dune faces with associated vegetation damage. At Mussel  
428 Point, for example, most of the ORV traffic utilises the intertidal beach area, and the increase in  
429 tracking in the new dune zone reflected a few vehicles (such as dune buggies) that were driven on  
430 sloping ground between the old dune toe and the new high tide line position. The tracking densities  
431 and trends measured at the monitoring sites were also similar to those observed elsewhere along  
432 the coastline (Figure 4). Greater increases in tracking density were observed at sites where there is a  
433 wider new dune zone that has become available for tracking. Reducing this footprint provides a  
434 focus for management to assist vegetation recovery and has potential to improve outcomes for  
435 native sand-binding species that are competing with more generalist exotics for the newly available  
436 area. Impacts on the growth of runners from existing native dune remnants were identified as a  
437 particular threat to be minimised where possible.



438

439 **Figure 8** Dune monitoring sites. Mussel Point site with the new dune zone establishing seaward of old dunes  
440 looking west (A), and east (B). Spinifex runners from old-growth native dunes north of Long Point (C), and at  
441 the Long Point monitoring site (D). New plant establishment at Aerial Beach (E). Vehicle tracking in the new  
442 dune zone at Long Point (F).

443

444

445

## 446 **4. Discussion**

### 447 **4.1 Management of prograding shorelines**

448 Tectonic uplift and isostatic adjustment from glacial de-loading are counterbalances to sea-level rise  
449 that will affect the speed and direction of coastal environment change (Church et al. 2013; Stammer  
450 et al. 2013). On the global stage, there are extensive coastal environments in seismically-active  
451 regions (Mogi 1974; Stern 2002), or subject to glacial isostasy (Peltier 2004; Shugar et al. 2014), and  
452 this suggests that a very considerable length of coastline is capable of progradation under conditions  
453 of uplift driven by these physical processes. Moreover, the contribution of uplift to shoreline  
454 evolution will remain important in the context of climate change since relative sea-level positions  
455 reflect the combination of vertical land mass motion and eustatic sea-level change (Cazenave &  
456 Llovel 2010). These factors contribute to variation in the magnitude and direction of relative sea-  
457 level changes at regional scales, with profound implications for day-to-day management (Nicholls &  
458 Cazenave 2010).

459 Erosion events and trends are commonly assessed as hazards with associated risks to assets  
460 and land-use (Crowell et al. 1999; Ferreira et al. 2006; Rabenold 2013), and the wealth of research  
461 effort associated with coastal erosion understandably reflects these concerns (Prasad & Kumar  
462 2014). Although shoreline progradation effects such as beach widening are intuitively associated  
463 with benefits, our results show that they may also generate downsides. A key learning from this  
464 study is the need for rapid appraisals of new 'opportunities' and any potentially negative impacts of  
465 progradation events, much the same as is commonly exercised in response to erosion and shoreline  
466 retreat. Therefore, responding to the dynamics of shorelines in a timely fashion is the key principle  
467 that emerges.

468 Where patterns of resource use and values have been established in response to relatively  
469 stable conditions, physical landscape changes may present risks to the continuation of those values  
470 in place. In this case, deleterious effects were generated from the exercise of existing land-use  
471 rights, with the landscape having changed in relation to them. By contrast, there will often be  
472 beneficial opportunities associated with the creation of new land where previous erosion or the  
473 encroachment of infrastructure has exerted a coastal 'squeeze' (Doody 2004; Orchard et al. 2020a;  
474 Tono & Chmura 2013). For example, a reversal of coastal squeeze effects has been previously  
475 reported from uplift in the 2010 Maule earthquake in Chile (Rodil et al. 2015) and 2010-2011  
476 Canterbury earthquake sequence in New Zealand (Orchard et al. 2020b). Additionally, sediment  
477 pulses originating from inland erosion and landslide events are known to exert pronounced effects

478 on coastline evolution due to influences on sediment budgets at various spatio-temporal scales, and  
479 these can also lead to progradation events (Carter et al. 1987; Syvitski & Milliman 2007).

480 Anthropogenic responses to landscape changes, such as those reported here, must be  
481 cognisant of the requirements of natural ecosystems and their need to constantly readjust and  
482 reassemble in response to landscape change (Berry et al. 2013). The identification of their spatial  
483 requirements is therefore a key component of effective coastal conservation (Martinez et al. 2014).  
484 Coastal zonation patterns demonstrate that adjustments to relative sea-level positions are  
485 obligatory for many habitats and ecosystem types – yet they may take time to fully manifest  
486 following physical landscape change (Turner et al. 1998). These lag effects make the detection of  
487 change more difficult, but can be addressed using techniques such as spatial modelling to predict  
488 longer-term impacts, and longitudinal studies to confirm actual response trajectories and the  
489 outcomes that result (Orchard et al. 2020b; Watts et al. 2020).

490 In this case study, examples of natural ecosystem responses to shoreline progradation  
491 included the formation of new foredunes seaward of their pre-quake position. Although the  
492 associated spatial requirements were initially unknown, they became clearly evident one to two  
493 years after the earthquake as new vegetation became established on the uplifted shore. The winter  
494 storms of 2017 appeared to have had an equilibrating effect on the position of a persistent  
495 vegetation line, and in most places throughout the study area this line has changed little since.  
496 Additionally, there has been increasing vegetation cover landward of this line which now forms the  
497 new dune toe (Table 3). Future changes not directly observable within the time frame of this study  
498 may include the landward boundary of the dune system becoming more stable as the new foredune  
499 intercepts sand supply. The likely consequence is progressive invasion of the sand-binding species by  
500 other terrestrial vegetation types. Consequently, the beach widening changes associated with  
501 tectonic uplift are expected to cause a seaward shift in the entire dune system over time. This  
502 illustrates principle of habitat migration in which, despite the nuances of lag effects, entire systems  
503 must move to new locations if they are to persist under conditions of change (Hovick et al. 2016;  
504 Schlacher et al. 2008b).

505 In the case of banded dotterel nesting habitat, there was an expansion of the sparsely  
506 vegetated substrates that are required for nesting success (Pierce 1983; Rebergen et al. 1998).  
507 However, plant community succession is also well underway on the uplifted gravel beaches (that  
508 support the majority of nesting habitat), in addition to the dune responses discussed above. This  
509 illustrates an interaction between vegetation recovery and nesting habitat, and in the absence of  
510 other changes is expected to progressively reduce the area of suitable habitat. In the meantime, the  
511 apparent bonus effect of additional habitat has been counterbalanced by an increase in ORV use



512 that poses a considerable threat to nesting (Figure 7). Similar ORV impacts have been reported in a  
513 study of hooded plover (*Charadrius rubricollis*) with a vehicle tracking density of 20% that resulted in  
514 an average of 6% of artificial nests being run over per day, implying an 81% loss within the  
515 incubation period (Buick & Paton 1989). In our case, these new human pressures are similarly un-  
516 managed at present, but were facilitated by physical landscape changes that rapidly altered the  
517 pattern of recreational activities rather than being the result of gradual access and user group  
518 changes that are typical of other studies (Luke & Schlacher 2008; Priskin 2003). In all likelihood, the  
519 new threats from ORV traffic have negated any potential reprieve from predation pressures  
520 conferred by the widening of high tide beaches, and demonstrate an important anthropogenic  
521 dimension for planning following natural disaster events.

522

#### 523 **4.2 Integrating recreation and conservation**

524 In theory, there is potential for win-win outcomes from the landscape changes reported here, if low  
525 impact options for ORV access could be identified using techniques such as designated routes. In  
526 practice, however, this may be difficult to achieve due to the widespread distribution of fragile  
527 vegetation types and shorebird nesting grounds that in many ways are a reflection of the high  
528 biodiversity values of this formerly-remote coast. This presents a conundrum for the design of fine-  
529 scale spatial planning approaches that might be used to weave a vehicle route through the new  
530 landscape now that the major physical barriers, such as headlands, are more easily bypassed. Three  
531 important dimensions for the identification of an integrated solution are discussed in the sections  
532 below, before summarising key learnings from this case in relation to the policy context.

##### 533 *Designated vehicle routes in intertidal areas*

534 Due to the distances that may be travelled in daily ORV excursions in our study area, a reliance on  
535 intertidal (e.g., low-tide) vehicle travel routes may not be an effective management solution for the  
536 protection of high tide beaches, since vehicle movements are likely to occur in those areas on return  
537 journeys. Additionally, there may also be impacts associated with ORV use in intertidal areas, since  
538 crushing effects have been reported for a wide range of sandy beach infauna (Lucrezi & Schlacher  
539 2010; Moss & McPhee 2006; Schlacher et al. 2008a; Schlacher et al. 2008c). New Zealand examples  
540 include impacts on shellfish such as toheroa (*Paphies ventricosa*) and tuatua (*Paphies*  
541 *subtriangulata*), and include sensitive intertidal elevation zones related to the distribution of  
542 different size-classes (Moller et al. 2014; Taylor et al. 2012; Williams et al. 2013). Consequently, the  
543 mapping of sensitive intertidal areas would be required to establish the degree of impact associated  
544 with ORV use and any potential low-impact options (Schlacher & Thompson 2007).



545

546 *Designated vehicle routes at the top of the beach*

547 Conceivably, a low-impact vehicle route could be found at the top of the beach if it represented a  
548 less sensitive environment. However, the degradation of old dune systems, that are characteristic of  
549 this location, is not a permissible activity due to their conservation status as endangered ecosystems  
550 (Holdaway et al. 2012). Aside from their existing conservation value they are crucial for progression  
551 of the current recovery process as the seed sources for new recruitment. Additionally, in the case of  
552 the primary sand-binding species *Spinifex sericeus*, repair of the dune system is facilitated by the  
553 vegetative growth of runners from the dune toe (Bergin 2008; Orchard 2014), and vehicle tracking in  
554 these areas are a further threat to recovery and seaward dune migration. Overall, the concept of a  
555 formed roadway in stable and less sensitive areas that may be found inland of coastal dunes may be  
556 appropriate when and where those conditions exist, but such a route would lie outside of the  
557 current dune system.

558

559 *Designated vehicle access areas for key activities such as boat launching*

560 An alternative set of options for the consideration of ORV access with appropriate environmental  
561 protection involves the identification of designated vehicle *areas* (being sections of coast where ORV  
562 use is permitted) rather than routes through sensitive areas. Such areas might also include a few  
563 'sacrificial' activity zones where high impact activities such as driving on dunes is permitted as a  
564 strategy to reduce impacts elsewhere; and these would logically be situated in areas where such  
565 impacts are already occurring. This line of thinking also applies to the continued use of vehicles at  
566 key sites for activities such as boat launching, especially where these activities are already occurring  
567 (and indeed may have a long history in some places). In relation to the previous management  
568 context, it is important to recognise that the post-disaster setting presents a new set of  
569 considerations that require active management to enable the continuation of such uses in an  
570 appropriate manner. These management responsibilities arise because of the increased potential for  
571 problematic ORV access into adjacent coastal areas associated with the landscape changes, leading  
572 to the need for formal controls or effective non-statutory interventions to ensure that such  
573 problems do not occur. Practically speaking, this means that coastal managers must designate  
574 vehicle access areas and their boundaries, in contrast to the pre-quake situation in which there were  
575 no formal provisions for appropriate locations and their use. While the contemplation of designated  
576 vehicle routes through sensitive areas depends heavily on the results of impact assessment work,  
577 the identification of vehicle access *areas* at key locations provides a tangible and important starting  
578 point.

### 579 **4.3 Responses and responsibilities under the policy context**

580 Impact assessments are a required aspect of the environmental policy cycle in New Zealand and  
581 similarly in other jurisdictions. For example, knowledge of impacts is typically necessary for day-to-  
582 day decisions on audits or permits functions, and in the review of statutory policies and plans at local  
583 through to national levels. A hierarchical arrangement of these measures is required by the Resource  
584 Management Act 1991 (RMA) which is New Zealand's primary environmental legislation alongside  
585 the Conservation Act 1987, Wildlife Act 1953, and Reserves Act 1977 (and amendments) (Memon &  
586 Perkins 2000). National Policy Statements prepared under the RMA are a key tool for environmental  
587 management that direct the preparation of more specific policies and plans within a devolved  
588 organisational framework (Memon 2002). Under the New Zealand Coastal Policy Statement 2010  
589 (NZCPS), requirements for the protection of coastal environments include avoiding adverse effects  
590 on indigenous species, ecosystems and vegetation types that are listed as threatened or at risk, or  
591 are naturally rare, and others that are at the limit of their natural range, or contain nationally  
592 significant examples of indigenous community types. Significant adverse effects must also be  
593 avoided in indigenous ecosystems and habitats that are only found in the coastal environment and  
594 are particularly vulnerable to modification, including estuaries, lagoons, coastal wetlands,  
595 dunelands, intertidal zones, rocky reef systems, eelgrass and saltmarsh; and similarly for areas of  
596 predominantly indigenous vegetation, and habitats that are important during vulnerable life stages,  
597 or for recreational, commercial, traditional or cultural purposes (Department of Conservation 2010).  
598 In practice, implementation of this statutory framework creates a substantial obligation for  
599 environmental planning informed by baseline and impact assessments (Orchard 2011). Management  
600 authorities, such as regional and district councils, are legally required to fulfil the statutory  
601 responsibilities set out in higher level legislation, creating a form of decentralised governance that is  
602 also common in other countries (UNDP 2004).

603 In addressing NZCPS requirements (and other legislation), methods to avoid the adverse  
604 effects of ORV use present a considerable challenge for management authorities. In remote settings  
605 where the enforcement is difficult, non-statutory measures that incentivise desirable driver  
606 behaviour are likely to be beneficial, alongside regulatory tools. For example, the support of user  
607 groups for spatial planning measures, such as permitted areas or vehicle routes, could play a self-  
608 reinforcing role by helping to socialise and generate buy-in for effective solutions. Conversely, there  
609 is an opportunity for user groups to develop initiatives around the design of such systems in advance  
610 of regulatory measures. Previous studies have shown that the success of motivational incentives also  
611 depends considerably on relationship-building between the public service, non-governmental  
612 organisations and local community stakeholders (Gunningham 2009; Koontz et al. 2004).

613           The establishment of areas that are closed to vehicles is one of the most obvious options for  
614 preventing their impacts on sensitive species and habitats. Moreover, beach closures at the location  
615 of shorebird nesting grounds have been shown to be effective in studies from Australia (Weston et  
616 al. 2012) and Namibia (Braby et al. 2001). In addition to reducing direct effects on nesting success,  
617 such as egg-crushing, protected areas can assist with reducing disturbance effects on essential  
618 wildlife behaviours such as incubation and foraging (Ruhlen et al. 2003; Weston & Elgar 2007). In  
619 combination, these considerations can inform decisions on the size and configuration of protected  
620 areas to optimise their efficacy and ensure they are well integrated with public access opportunities.  
621 These strategies, behaviours, and good practice examples can be incentivised by non-statutory  
622 motivational initiatives led by either NGOs or government authorities, and are a great focus for  
623 collaborative design and promotion. If such measures prove to be effective at reducing impacts  
624 where vehicle access is permitted, they will undoubtedly improve its consistency with established  
625 environmental objectives.

626

#### 627 **4.4 How can beach users reduce their impacts?**

628 While management authorities must respond to their statutory obligations under relevant  
629 legislation, there are certainly voluntary actions available to beach users to reduce their actual and  
630 potential impacts. They include, perhaps most importantly, the use of existing vehicle tracks where  
631 they exist, and the exercise of caution or reconsideration if there are no such well-formed tracks.  
632 When travelling in environmentally sensitive areas, ORV users can take practical steps such as  
633 looking out for wildlife, maintaining an appropriate separation distance, taking note of fragile  
634 vegetation and habitat types, and keeping an eye out for signage and various types of enclosures  
635 that can be used to indicate their presence. Driving speeds and setback distances from wildlife are  
636 known to be important determinants of the degree of disturbance associated with motorised  
637 vehicles (Schlacher et al. 2013). By extension, other characteristics of vehicle types, such as the level  
638 of noise they emit, are also likely to contribute to the adverse impacts of vehicle use in particular  
639 situations. Conversely, this presents opportunities for vehicle users to select their transport mode  
640 purposefully to avoid impacts and gain the acceptance of other stakeholders. In the near future, for  
641 example, electric bikes and other electric vehicle types may facilitate less invasive access modes in  
642 sensitive wildlife habitats, improving the options for sustainable travel in these areas. Ultimately a  
643 consistent focus on recreational impacts could be applied to all access modes to help manifest a  
644 light-footed, low impact relationship with the natural environment while encouraging visitation by  
645 people.

646

#### 647 **4.4 Concluding remarks**

648 This study highlights the need to provide for natural environments in responses to landscape  
649 changes. Failure to do so is likely to undermine conservation objectives with potentially drastic  
650 consequences. Even small changes in critical parameters can lead to new impacts over large areas.  
651 Recovery actions following disturbances must pay attention to impacts from both the original  
652 disturbance and consequential changes, and identify trade-offs between desirable objectives. This  
653 study illustrates these requirements in a situation of shoreline progradation that offered new land-  
654 use opportunities for various forms of nature-based recreation and access. The recreation-  
655 conservation nexus was a key driver of change and negative impacts were generated by the  
656 interaction between access opportunities and dynamic natural environments. A key lesson that  
657 emerges is the need for timely impact assessments across the social-ecological spectrum whenever  
658 physical landscape changes alter the accessibility of geographical locations and resources. Similar  
659 principles are transferable to many other natural hazard and disaster recovery settings and will help  
660 to promote the achievement of sustainable development and resource management goals.

661

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673

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