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Attrition of recommended areas for protection: clearance of ecologically significant vegetation on private land

Adrian Monks[1*], Ella Hayman[2] and Susan Walker[1]

[1]Manaaki Whenua – Landcare Research, Private Bag 1930, Dunedin 9054, New Zealand
[2] Manaaki Whenua – Landcare Research, Private Bag 11052, Palmerston North 4442, New Zealand
[*]Author for correspondence (Email: monksa@landcareresearch.co.nz)

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Abstract: The area of indigenous vegetation and habitat remaining on New Zealand's primary agricultural lands continues to decrease, but it has been difficult to obtain reliable estimates of the extent and causes of loss. We assess change and identify predictors of vegetation clearance in 856 recommended areas for protection (RAPs) from 35 ecological districts in the North and South Islands, New Zealand, for the period 1989 to 2015. Over 27 years, 7152 ha of these RAPs were cleared (2.3% by area), with rates varying over space and time. Native forest was least commonly cleared (422 ha removed), followed by native non-woody vegetation (1294 ha), native shrubland (1378 ha), and 'other' vegetation (4058 ha). The probability of clearance peaked during 2001 to 2008 at 0.14% yr^{-1}, but it was still nearly double the 1989–2001 levels (0.06% yr^{-1}) from 2008 to 2015 (0.11% yr^{-1}). The annualised clearance probabilities after 2001 were comparable to the rates of deforestation in the pre-1840 period of human settlement and about a third of that recorded from 1840 to 1970, the most intensive known period of anthropogenic clearance in New Zealand. Clearance rates were higher around the edges of small RAPs without legal protection and in drier, cooler areas, generally and increasingly over time. Amount of surrounding cropping/horticulture was negatively associated with clearance, as initially was dairy before developing a slightly positive association. Forestry was positively associated with clearance up until 2008. Our results show proportionally greater clearance of marginal agricultural land with high biodiversity values as time goes on, probably facilitated by the increasing use of technology, such as irrigation and fertilisation, to circumvent environmental limitations to plant growth. These results demonstrate ongoing attrition of the highest-value native habitat remaining on private land, and the inadequacy of the current protection framework to safeguard it.

Keywords: biodiversity loss, disturbance, habitat destruction, protected natural area programme, remote sensing, vegetation clearance

Introduction

The native vegetation that supports indigenous plant and animal species continues to decline in extent across New Zealand's primary agricultural lands. From 1996 to 2012, 31 000 ha of tussock grassland, 24 000 ha of indigenous shrubland, and 16 000 ha of indigenous forest were cleared across New Zealand (MfE & Stats NZ 2018), in addition to many other areas of habitat for native species. Continued loss represents an ongoing threat to New Zealand's remaining biodiversity (Ewers & Didham 2008; Innes et al. 2010; Holdaway et al. 2012) and will probably have implications beyond the current patterns of loss as functionally extinct native remnants senesce ('extinction debt'; Tilman et al. 1994).

Human-induced vegetation change and habitat destruction have transformed the New Zealand landscape. In the 560 or so years between Polynesian settlement and the time of early European settlement (1840–1860), human-caused fire reduced forest cover by nearly 50% compared to pre-human times, leaving behind a mosaic of bracken fernland, grassland and shrubland (McGlone 1989; Wardle 1991). Most of this change occurred in the drier, eastern parts of New Zealand, especially of the South Island (McGlone 1989). European settlement brought strong demand for indigenous timber and flax, and the advent of pastoralism initiated a tsunami of plant and animal naturalisations (Wardle 1991).

Pastoral development was initially limited to the most arable areas (the flatter, more fertile and warmer lowland), but later programmes (such as the land clearance subsidies of the 1970s and early 1980s) saw increasingly marginal lands cleared for agriculture (MacLeod & Moller 2006). The combined effects were sufficient to see most gently sloping land at low elevations cleared for agriculture or urban use, as well as much of the lower-elevation hill country. In recent decades, the development of irrigation infrastructure and the transition to high-input agriculture (MacLeod & Moller 2006) mean that much previously marginal land is now economic to develop, heralding a fresh wave of threats to remnant native

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vegetation on private land. As a result of all these pressures, natural ecosystems at low to moderate elevations in drier and warmer environments are greatly reduced from their former extent and remain at risk of clearance (Cieraad et al. 2015; Weeks et al. 2013).

Thirty-two percent of New Zealand's total land area is currently managed by the Department of Conservation (DOC) (MfE & Stats NZ 2018). However, land protected for conservation purposes was mostly not initially suitable for agriculture, and so mountainous regions are over-represented and lowland areas are poorly represented (Cieraad et al. 2015). This pattern has continued through the tenure review process, which has overseen the retirement of thousands of hectares of Crown pastoral lease into public conservation lands, mostly at higher elevations (Brower et al. 2010; Cieraad et al. 2015) and the privatisation and often consequent subdivision and development of land at lower elevations (Walker et al. 2008a; Brower & Page 2017). As a result, indigenous cover that now remains on low-elevation private land is critical to landscape-scale ecological integrity. In some cases, the last representatives of many ecosystems are only found on private land (Holdaway et al. 2012; Brown et al. 2015). In recognition of this importance, many landowners have sought to protect their native remnants through covenants with the QEII Trust, and Ngā Whenua Rāhui covenants registered through DOC. Community-led landscape-scale initiatives such as Cape to City[1] and the Beyond Orokonui project[2] emphasise the contribution of isolated patches in a linked-up ecological landscape to maintaining indigenous biota. Some district councils now have rules constraining the clearance of native vegetation, although the criteria vary widely between councils (e.g. Myers et al. 2013).

However, the success of conservation initiatives on private land is piecemeal. Loss of indigenous vegetation continues, through either wilful clearance or ecological ignorance, at both the private landowner and the enforcement agency level. Sometimes knowledge of these events finds its way into the public domain, such as the 2018 clearance of extensive areas of the nationally endangered shrub *Muehlenbeckia astonii* on Kaitorete Spit[3]. More often the change has been insidious, being only identified during broad-scale stock-taking exercises such as Ministry for the Environment reporting (MfE & Stats NZ 2018) or the threatened environments classification (Cieraad et al. 2015).

The New Zealand Biodiversity Strategy goal of arresting loss (DOC & MfE 2000) requires more detailed information on the threats to habitat of indigenous species. We need to understand where and why loss is occurring in order to aid the planning, advocacy and prioritisation of efforts to secure the remaining biodiversity in these landscapes. Land uses continue to evolve, and a key need is to understand what those changes might mean for the habitat security of the indigenous biota. The threatened environment classification (Cieraad et al. 2015) provides a coarse-grained starting point for identifying where the bulk of historical clearance has taken place, but the thematic and spatial resolutions of the national Land Cover Database (LCDB) are too low to provide an adequate basis for describing on-the-ground changes in indigenous cover and habitat for indigenous species (Cieraad et al. 2015; Dymond et al. 2017). A natural expectation based on historical patterns suggests that increasing scarcity of prime agricultural land will lead to a shift in development to more marginal lands that are either less productive or require higher inputs to maintain productivity, as shown by Weeks et al. (2013) for indigenous grasslands.

Here we seek to identify the fine-grain factors associated with vegetation clearance from 1989 to 2015 and whether these have changed over time as the most suitable agricultural land is used up or with economic cycles favouring particular kinds of agricultural investment. We expected increasing rates of clearance of more marginal land during this period. Our analysis used Protected Natural Areas Programme (PNAP) surveys from the previous 34 years and satellite imagery from the last 26 years to identify where and why loss is occurring. PNAP surveys were designed to identify, and prioritise for protection 'representative samples of ecosystems, landforms and scenery' (Kelly & Park 1986) that were not already part of the public conservation land network in order to meet the requirements of section 3(1)(b) of the Reserves Act 1977 (Walker et al. 2008b). The surveys encompassed private and public land and still represent the most comprehensive, publicly available surveys of significant areas of native biodiversity on private land in New Zealand (Bellingham 2001). Each survey identified and mapped locations of recommended areas for protection (RAPs), and because intactness was a founding principle of the ranking scheme, RAPs can be considered the most ecologically valuable of the unprotected native vegetation and habitat important for native wildlife that remained at the time (Walker et al. 2008b). As such, mapped RAPs represent a sound basis for identifying areas that undoubtedly are, or were, ecologically significant indigenous vegetation and/or habitats of native species.

Our first expectation was that formal legal protection of the RAP would substantially reduce the risk of clearance (H1). Formal legal protection includes covenanted land, or land subsequently reserved by local or central government, but excludes general protection afforded by clearance rules in district plans. We also expected environmental variables, surrounding land use, and characteristics of the indigenous fragments themselves to determine the likelihood of vegetation clearance. We predicted that small RAPs would be much more likely to be cleared than larger ones (H2) because they are easier to clear for particular projects or below permitted thresholds in district plans. We expected land with higher economic development potential to experience higher probability of clearance. Hence we predicted the highest likelihoods of recent clearance to be on more arable land, as indicated by physical characteristics (low elevation, shallower slope, lower soil moisture deficit and less variable rainfall) or surrounding land use (high dairying and cropping/horticulture; H3), or on sites most suitable for exotic forestry (H4). We specifically test for effects of exotic forestry because this has been previously − and controversially − identified as a cause of vegetation clearance (Walker et al. 2006; but see Brockerhoff et al. 2008). We also predicted greater risk of clearance in more densely settled areas due to higher pressure for more intensive use of land (H5). Our expectation was that RAPs generally would be more likely to suffer loss around the margins than total or partial clearance (H6). We expected there to be less recognition of the ecological value of low-stature vegetation (grassland/herbfields and shrubland) than forest, with probability of loss varying accordingly (H7). Finally, with new investments in higher-intensity agriculture and irrigation in many parts of

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1 https://www.capetocity.co.nz/; accessed 11 June 2018
2 https://www.beyondorokonui.org.nz/; accessed 11 June 2018
3 https://www.stuff.co.nz/environment/103507786/farmer-destroyed-a-third-of-countrys-naturallygrowing-endangered-shrubby-tororaro-plants; accessed 15 May 2018

New Zealand, we expected that, over time, the relationship between biophysical indicators and risk to RAPs would relax as technology overcame the economic constraints of farming marginal lands (H8). Moreover, we expected the probability of clearance to vary over time, reflecting economic cycles in agriculture. For example, MacLeod and Moller (2006) show that the ending of farm subsidies in the early 1980s halted and slightly reversed the amount of land in agricultural production in the following decade.

Methods

Data

We used publicly available digitised maps of RAPs from PNAP surveys to identify areas of ecologically significant vegetation. While primarily native, this vegetation may also be exotic or contain exotic components because RAPs were also identified on the basis of being habitat for native fauna. We initially prioritised surveys where field work was undertaken between 1984 and 2002 so that we could examine change over more than one time period for each ecological district using RAPs identified in near time to the respective periods. Thereafter we added digitised surveys with earlier field work as resourcing allowed. Using the relevant PNAP survey reports (see Table S1 in Supplementary Material), we augmented the attribute tables with additional information, including ecological region and district. There were 856 RAPs spread across 1163 individual vegetation fragments in our data set. RAPs ranged from 0.1 to 8066 ha, covering 296 479 ha from areas in both North and South Islands (Fig. 1). While not exhaustive in coverage of the full environment range across New Zealand, there is nonetheless considerable environmental variation represented by these RAPs.

We assessed clearance of existing RAP vegetation over time in the RAPs using satellite imagery, with a Red-Green-Blue composite of near infrared (NIR), shortwave infrared (SWIR) and red (R) and a stretch of 2 standard deviations. This combination was chosen to maximise discrimination of vegetation type (SWIR) and condition (NIR), with the red band the least responsive to atmospheric differences between imagery sets. The source imagery used depended on the period over which vegetation change was being assessed. We used Landsat 4 (1988–1993 at 4–15 m resolution); Landsat 7 (2001–2002 at 7–15 m resolution); Spot 5 (2006–2008 and 2011–2013 at 5–10 m resolution); and Landsat 8 (2014–2015 at 8–15 m resolution). Composite cloud-free images were available from the Ministry for the Environment. Google Earth imagery and Spotmaps (2008–2009 and 2013–2014 in true colour) were also used for clarification when needed (Fig. 2).

Vegetation clearance for each RAP was assessed visually by manually comparing images using the MapAccuracy software (version 0.9.2; unpublished Manaaki Whenua – Landcare Research, based on TuiView; Clewley et al. 2014) which allows multiple imagery sets for the same point to be viewed simultaneously. Where a RAP contained multiple polygons, each polygon was assessed individually. The change resulting from the clearance was classified according to the categories in Table 1. Change due to vegetation succession was not classified as a change for this analysis. In some cases, existing non-native vegetation types already in agricultural production were included within RAP boundaries, either in error due to inaccurate digitising of RAP boundaries, or deliberately. Some of these probably underwent change such as ploughing and harvesting. Due to the difficulty of separating pasture

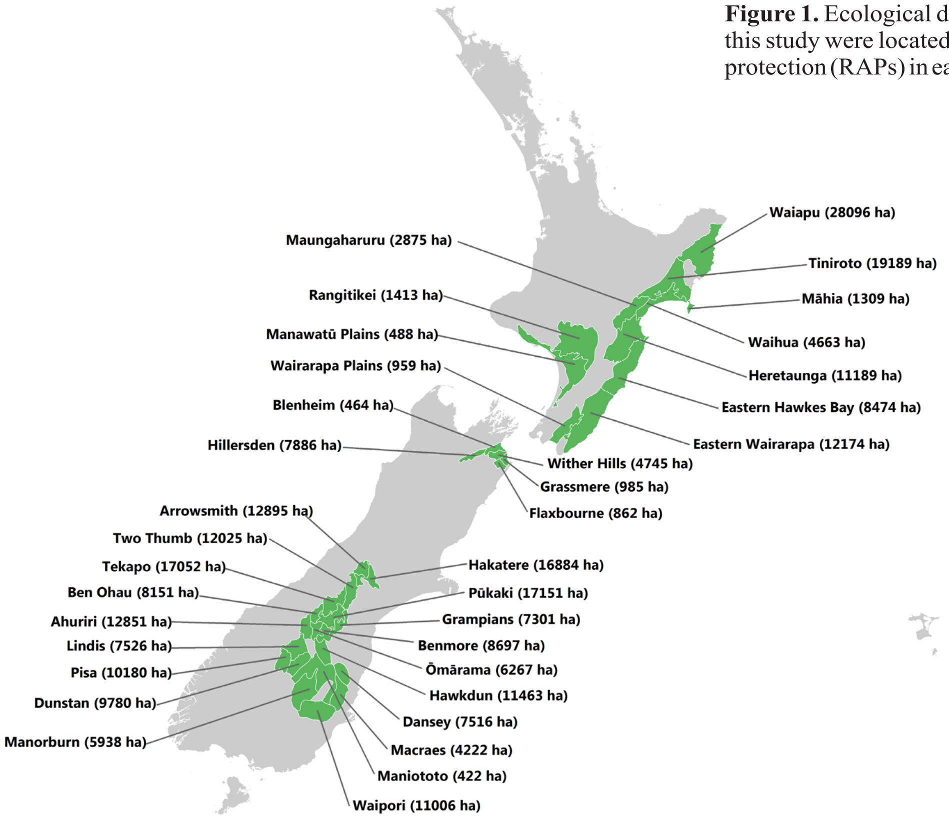

Figure 1. Ecological districts in which the PNAP surveys used in this study were located. Total areas of the recommended areas for protection (RAPs) in each ecological district are given in brackets.

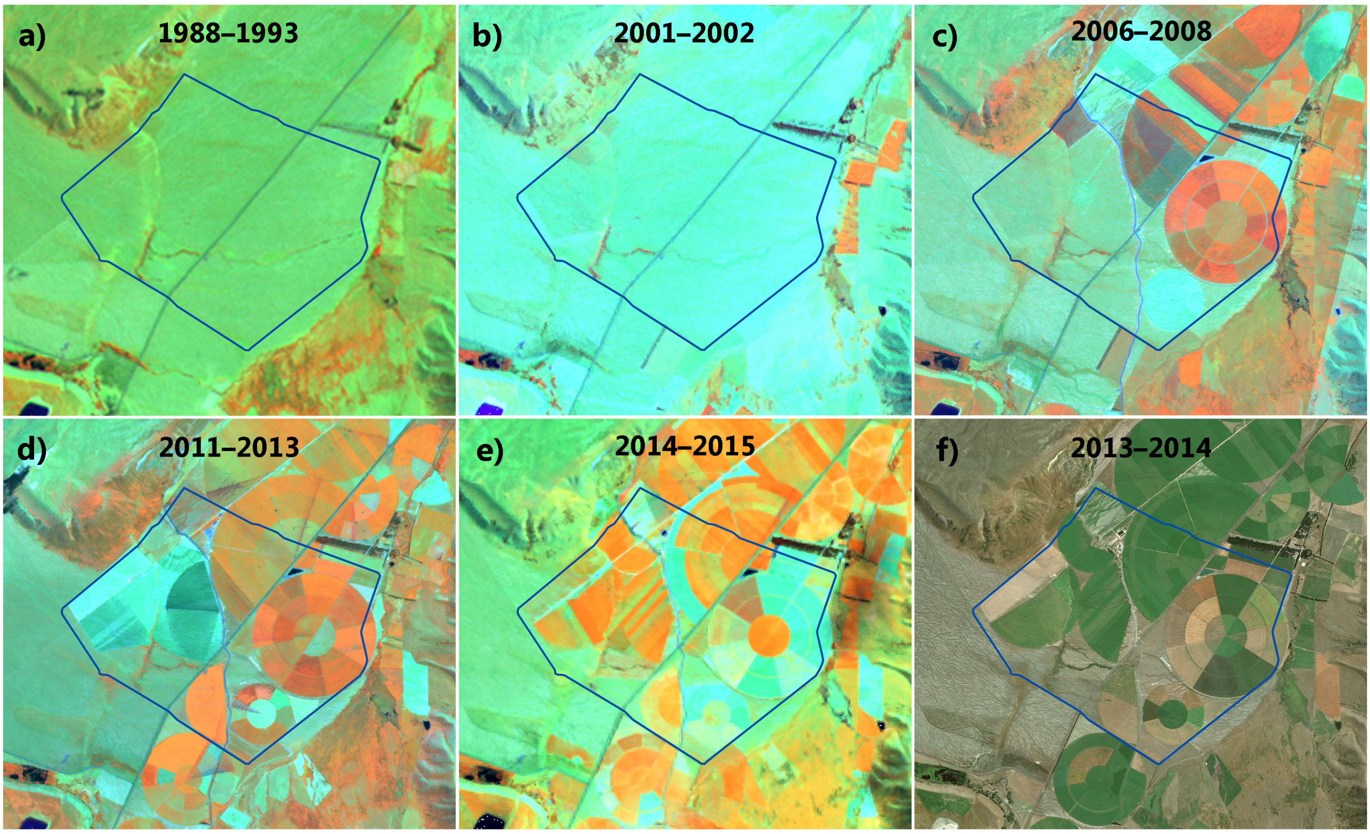

Figure 2. Composite image of a site at Spring Creek, Mackenzie Basin, for different imagery sets at different times: (a) Landsat 4 imagery (1988–1993); (b) Landsat 7 imagery (2001–2002); (c) Spot 5 (2006–2008); (d) Spot 5 (2011–2013); (e) Landsat 8 (2014–2015); (f) Spotmap in true colour (2013–2014). The blue boundary shows the boundary of the RAP. The original vegetation (as classified by LCDB1), shown intact up to and including 2001–2002 (a, b), was 97% 'depleted grassland', 2% 'low producing grassland', and was classified as 99% 'native non-woody' and 1% 'other' using our revised classification.

Table 1. Categories used to characterise clearance of native vegetation in recommended areas for protections (RAPs).

Category	Description
No change	No change detectable
Clearance	Clearance of vegetation only, through spraying, ploughing, etc.
Exotic plantation	Afforestation with plantation forestry
Water reduction	Reduction in extent of water for a wetland
Track	Tracks and roads
Building	Erection of a building
Other	E.g. quarries, mines, exclosure fences, orchards, lifestyle blocks/houses, canal / water race, ponds

from native herbaceous cover at these resolutions, we were only able to identify such cases for harvested forestry. These cases (180 polygons) were omitted from the analysis. Change was assessed for three periods dictated by the timing of the surveys and the date of the imagery: 1989–2001 (period 1), 2001–2008 (period 2), and 2008–2015 (period 3). Due to the historical nature of these changes we were unable to ground-truth our assessments of change. However, by using multiple image sources, and manually evaluating change for each RAP polygon, we expected to minimise any bias in accuracy with respect to the relevant predictor gradients.

RAP polygons often spanned environmental gradients, and the attributes of a location on one side of a polygon could be quite different to those on the other side. To address this heterogeneity we resampled the polygons at multiple points using a generalised, random-tessellation, stratified survey design (using the package spsurvey for R; Kincaid & Olsen 2016). An equal probability design was used, with the number of points in each polygon determined by polygon area. Sampling resolution was set at 1 per ha because most of the explanatory variables were also measured on a 1 ha scale. The resampled points represented an unbiased sample from change and non-change polygons, with some but not all polygons with an area <1 ha containing a random point. Parts of RAPs where vegetation clearance was recorded were not considered when assessing change in subsequent time periods.

For each point we extracted a number of spatial measures that related to our hypotheses using GIS tools. Protection status was sourced from Cieraad et al. (2015). Average slope within a 200 m radius was calculated from a 25 m slope layer. A proxy for the distance to the edge of the fragment was calculated from the Land Cover Database (LCDB4; Pairman 2014) as the distance to the nearest non-native land cover class. We used the LCDB4 reclassified 1996/97 imagery for the first period, 2001/02 imagery for the second, and 2008/09 imagery for the third. Penman rainfall deficit and mean annual temperature were extracted from the Land Environments of New Zealand underlying data layers (Leathwick et al. 2002). Annual rainfall variability was extracted from a layer constructed from fitting a spline to annual rainfall statistics from 305 rain gauges (Manaaki Whenua – Landcare Research, unpublished). The measure for each cell in this rainfall layer is the estimated coefficient of variation in mean annual rainfall. We calculated a proxy for level of development as the number of pixels occupied by roads within 1 km of each point. Road location data were sourced from the Land Information New Zealand NZ Road Centrelines (Topo, 1:50 000) layer.[4] The area of land in dairy, cropping and forestry within 1 km of each point was calculated from the AgriBase national spatial farms database (AsureQuality 2011). We reclassified the New Zealand Landcover Database (LCDB4; Pairman 2014) into four vegetation types to be used as predictors: 'native forest', 'native shrubland', 'native non-woody vegetation' and 'other' (Table S2). Correlations between all predictors were less than 0.63.

Analysis

We modelled point-level change as a binary variable using a generalised linear mixed model. Errors were assumed binomial distributed, with a logit link function. We included unique RAP and (within RAP) fragment identity as random intercept terms to account for lack of independence between points from the same RAP or fragment. Fixed effects were average slope; distance to nearest exotic cover class; density of roads within 1 km radius; variation in mean annual rainfall; mean annual Penman water deficit; mean annual temperature; land area of the RAP at the start of the time period; land areas in forestry, dairy and horticulture/cropping within a 1 km radius; time period; and vegetation type. Interaction terms between time period and all fixed effect terms were also included in the model.

The models were fitted using Hamiltonian MCMC implemented in the package Stan accessed through the R package rstan (Version 2.15.0; Stan Development Team 2017). All numerical predictors were centred and scaled to have a mean of 0 and unit standard deviation. We assumed N(0, 20) priors for all fixed effects. Random effects assumed normal priors, with a mean of 0 and the standard deviation an estimated parameter. We assumed weakly informative Gamma(3, 1.25) priors on the standard deviations of the random effects. Burn-in was 200 iterations, and inference was based on 1000 posterior samples thinned by four for each of four MCMC chains. Model convergence was assessed using trace plots and the Gelman–Rubin convergence diagnostic R-hat (Gelman et al. 2004). For inference we calculated medians and 95% credible intervals (presented as 'median [95%CI]'). However, because our points are an arbitrarily large sample from the landscape, the widths of the credible intervals are less important to interpretation than the median size of the effects. For most results we present a main effect for the parameter of interest (slope for numerical variables, and a mean for categorical variables) and then a difference from this mean for each time period (Δ_2 and Δ_3 for periods 2 and 3, respectively). For graphical presentation of the modelled effects, we calculated effects for each period by focal variable combination, averaged over the other terms in the model. Uncertainty was estimated by performing this calculation for each of the posterior samples of the parameter estimates and calculating medians and 95% credible intervals for the calculated effect for each value of the variables of interest. Annualised probability of clearance was calculated as $1–p^{1/N}$, where p is the probability of persistence over the measurement period and N is the length of the measurement period.

Results

Over the 27-year period (1989–2015) covered by this study, 7152 ha out of a maximum potential 296 479 ha of land identified as RAPs (2.3%) was cleared of its existing vegetation cover. This loss was uneven over the periods, with 1915 ha cleared from 1989 to 2001 (160 ha yr^{-1}), 2830 ha from 2001 to 2008 (404 ha yr^{-1}), and 2206 ha from 2008 to 2015 (315 ha y^{-1}). This resulted in annualised probabilities of loss of remaining RAP vegetation of 0.06% yr^{-1} (1989–2001), 0.14% yr^{-1} (2001–2008) and 0.11% yr^{-1} (2008–2015) across ecological districts in the study.

However, there were clear differences in loss rates between districts (Fig. 3). For example, very high rates of loss were recorded in Māniototo in the second time period (13.7% yr^{-1}), but no loss in any period for the more mountainous regions such as Ahuriri, Arrowsmith, Ben Ohau, Grampians and Hawkdun ecological districts. The patterns highlight a shift in the focus of new agricultural development from the East Coast region of the North Island from 1989 to 2001, to the inland basins of Canterbury and Otago from 2001 to 2015 (Fig. 3a–c). The risk to vegetation in the North Island remains moderate throughout, with no ecological districts free from clearance. Peaks in the South Island are associated with a 2001–2015 development pulse (Fig. 3d–f), and contrast with the several high-altitude RAPs that had no recorded clearance. Native forest was cleared the least (1.99% of its cover; 422 ha), followed by native non-woody vegetation (1.5%; 1294 ha), native shrubland (4.2%; 1378 ha), with 'other' vegetation cleared the most (9.3%; 4058 ha).

Modelled predictors of vegetation change indicated substantial shifts in the drivers of vegetation loss over time. The largest effects were for time period, initial area, legal protection, distance to edge, mean annual temperature, and vegetation type (Fig. 4). Broadly, there was a trend for increasing clearance through time, despite the first period being 5 years longer than subsequent periods (time period main effects Δ_2 2.32 [1.78–2.89]; Δ_3 4.05 [3.46–4.64]). However, this general trend was extensively modified according to the location and circumstance of the RAP. Legal protection was very effective at reducing loss (H1), with lower rates of loss across all time periods (0.01% clearance on protected vs. 2.4% on non-protected). The effectiveness of legal protection increased in the second and third periods, with the largest relative benefit of protection evident from 2001 to 2008 (Fig. 4a; main effect −2.39 [−3.05 to −1.83]; Δ_2 −2.71 [−4.18 to −1.51]; Δ_3 −0.91 [−1.67 to −0.05]).

In keeping with expectation (H2), small patches were significantly more likely to be reduced in size by clearance than

[4] https://data.linz.govt.nz/layer/50329-nz-road-centrelines-topo-150k/; accessed 10 May 2018.

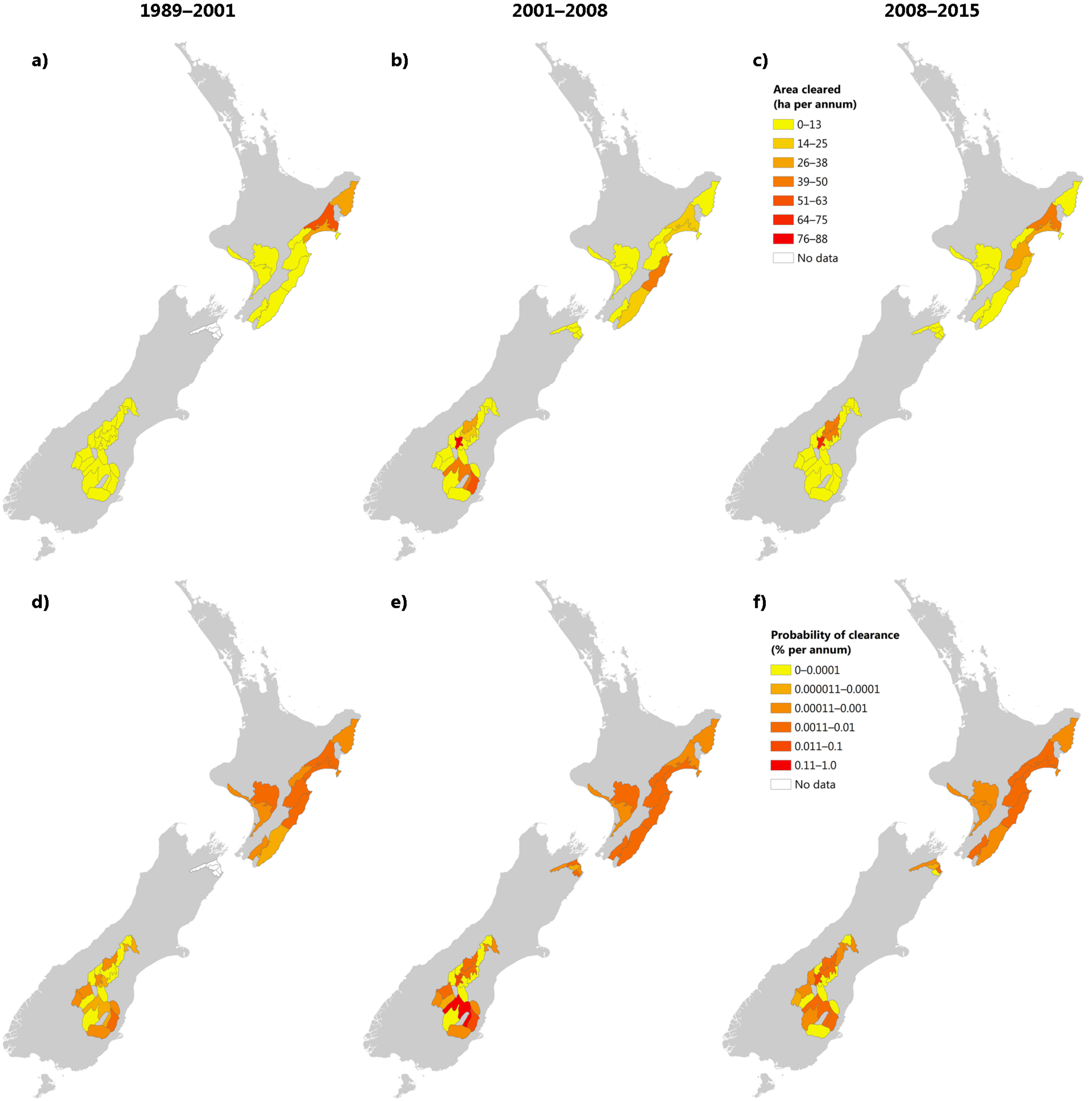

Figure 3. Annualised land clearance, by time period and ecological district. Quantities have been summed across RAPs and converted to annual clearances. Clearance is presented as the mean amount in hectares cleared per annum (a–c) and the annualised probability of clearance (d–e). Grey indicates the land area of New Zealand not included in this study.

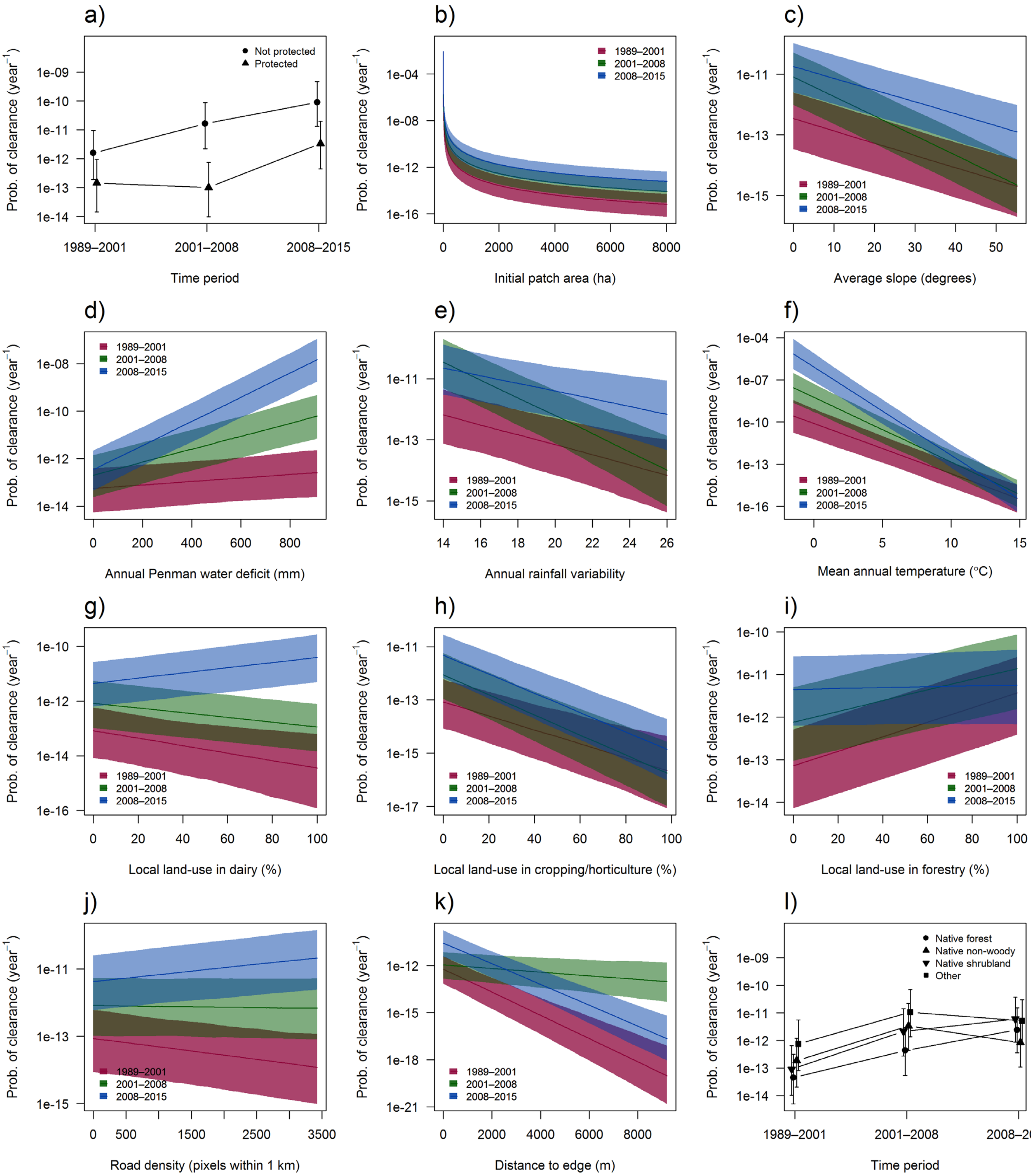

Figure 4. Modelled effects of the predictors on the annualised probability of clearance for each time period. All predicted effects are plotted on the log scale and show the modelled relationship for each variable, when all other variables are fixed at their average value. The effects plotted are: (a) legal protection; (b) initial area of RAP polygon; (c) average slope of surrounding land in 200 m radius; (d) mean annual Penman water deficit; (e) coefficient of variation of the annual rainfall; (f) mean annual temperature; (g) percentage of the local land area (within 1 km radius) in dairy; (h) percentage of the local land area used for cropping or horticulture; (i) percentage of the local land area in forestry; (j) local road density (within 1 km radius); (k) distance to edge of native vegetation fragment, as estimated by distance to nearest non-native LCDB pixel; (l) vegetation type.

large patches (Fig. 4b; main effect −3.96 [−4.17 to −3.77]). Although there was a significant trend for initial patch size to become less important as time went on (Δ_2 0.17 [0.07–0.26]; Δ_3 0.43 [0.33–0.53]), these differences were minor compared to the overall main trend.

There was significant model support for the hypothesis that the greatest clearance of RAP vegetation occurs in more arable areas (H3). The steepest areas were least likely to be cleared (Fig. 4c; main effect −1.01 [−1.16 to −0.87]). The increased clearance probability between 1989–2001 and 2001–2008 was mostly experienced in flatter areas, whereas clearance in 2008–2015 was more likely in steeper areas compared to period 2 (Δ_2 −0.6 [−0.78 to −0.42]; Δ_3 0.03 [−0.14–0.23]). While in 1989–2001 drier areas (higher penman deficit) were only slightly more likely to be cleared than wetter areas (Fig. 4d; main effect 0.35 [0.04–0.64]), there was an increasing trend over time for drier areas to become much more likely to be cleared (Δ_2 0.9 [0.75–1.07]; Δ_3 1.97 [1.82–2.13]). Sites with more variable rainfall were slightly less likely to be cleared than less variable sites (Fig. 4e; main effect −0.58 [−0.88 to −0.28]). Increases in clearance between 1989–2001 and 2001–2008 were mostly areas with less variable rainfall (Δ_2 −0.46 [−0.53 to −0.39]), whereas change was spread over a wider range of rainfall variability in 2008–2015 (Δ_3 0.14 [0.06–0.21]). Overall, cooler areas were more likely to be cleared than warmer areas, and this trend became increasingly pronounced as time went on in the study (Fig. 4f; main effect −2.56 [−3.06 to −2.1]; Δ_2 −0.72 [−0.92 to −0.53]; Δ_3 −1.92 [−2.13 to −1.72]).

In terms of the effect of the predominant local land use, patterns varied between agri-industries. In the 1989–2001 period, areas with a high proportion of local land in dairy were the least likely to exhibit land clearance (Fig. 4g; main effect −0.14 [−0.24 to −0.05]). However, this pattern slowly changed over time, and by the final period more clearance was occurring in areas with more dairy farming than areas without much dairy farming (Δ_2 0.05 [−0.04–0.16]; Δ_3 0.24 [0.15–0.34]). For cropping/horticulture, most clearance came in areas without extensive cropping (Fig. 4h; main effect −0.3 [−0.42 to −0.19]), and this pattern became more pronounced from 2001 onwards (Δ_2 −0.13 [−0.24 to −0.02]; Δ_3 −0.11 [−0.21–0]). Our expectation (H4) that RAPs with extensive areas of surrounding forestry would be more likely to be cleared than those with no nearby forestry was supported in the periods from 1989 to 2008 (Fig. 4i; main effect 0.38 [0.34–0.42]; Δ_2 −0.1 [−0.14 to −0.06]). However, by the 2008–2015 period, local forestry did not predict clearance (Δ_3 −0.36 [−0.41 to −0.3]).

In contrast to our predictions (H5), there was a weak trend for higher clearance in more remote areas during 1989 to 2001 (Fig. 4j; main effect −0.15 [−0.22 to −0.07]), which had disappeared by 2001–2008 (Δ_2 0.14 [0.05–0.23]). By 2008–2015 there was a weak positive association between road density and probability of clearance (Δ_3 0.27 [0.18–0.35]). As expected (H6), vegetation clearance was most likely to occur on edges abutting exotic vegetation compared to points deep within fragments (Fig. 4k; main effect −2.8 [−3.4 to −2.11]). However, this effect was very weak in 2001–2008 (Δ_2 2.36 [1.7–3.05]), suggesting clearance of more expansive areas within RAPs during the period. By 2008–2015 clearance around edges was again more common (Δ_3 0.29 [−0.5–1.05]).

All vegetation types were more likely to be cleared in 2001–2008 compared to 1989–2001 (Fig. 4k). Native forest was increasingly likely to be cleared over time in successive periods, but generally at lower levels than other vegetation types, consistent with predictions (H7). Native shrubland followed a similar trend to forest, going from third mostly likely to be cleared to equal most likely with 'other' by 2008–2015 ('native shrubland' main effect 0.73 [0.52–0.96]; Δ_2 0.86 [0.51–1.2]; Δ_3 0.19 [−0.08–0.47]). The 'other vegetation' category was the most likely to be cleared in 1989–2001 and 2001–2008 ('other' main effect 2.83 [2.63–3.05]; Δ_2 0.36 [0.02–0.67]), but clearance declined to similar levels to native shrubland by 2008–2015 (Δ_3 −2.1 [−2.4 to −1.8]). Clearance of native non-woody vegetation declined between 2001–2008 and 2008–2015 (Δ_3 −2.55 [−3.13 to −2]).

Discussion

Our data display a pattern consistent with native vegetation and habitat for native biodiversity on private land being under increasing pressure from development between 1989 and 2015, and legal protection being a strong, albeit imperfect, guarantor of security for native vegetation. Lands that were more marginal for agri-development were more likely to be cleared by the end of the study period than at the beginning, though at lower rates than less marginal lands, consistent with predictions (H8). The most important explanatory variables were time period, legal protection, initial patch area, mean annual temperature, distance to edge, Penman deficit, and vegetation type.

Legal protection prevented loss (H1) and was increasingly effective over time compared to unprotected lands. Nevertheless, some clearance on protected land was observed, mostly by ploughing or spraying and on the scale of a few hectares (up to 13 ha in a single RAP). One instance involved the installation of a predator-proof fence (1.7 ha at Redbank, Macraes Flat), but in most cases loss represented clearance for agriculture or forestry (38 ha over entire study). We do not have dates for the commencement of protection, and some of the observed clearance may have occurred prior to formal protection being achieved. However, it seems unlikely that land that was cleared and put into production would subsequently be reserved or covenanted.

Consistent with our expectations, small patches were more vulnerable to clearance than large patches (H2), and clearance from the edge (H5) of larger fragments was more common than clearance of the core. Both forms of loss are incremental and the type of change least likely to be noticed by regulators. Moreover, edges and small fragments are likely to have reduced native dominance compared to the interior of large tracts of vegetation (Brothers & Spingarn 1992; Ecroyd & Brockerhoff 2005), and therefore may not be perceived as particularly valuable, or even a nuisance for some farming operations. However, the effect of gradual attrition of the edges of larger fragments, exposing more of the interior to edge effects, is an inevitable slow decline of the ecological integrity of larger tracts, which are essential for conserving many biotic groups. Ewers and Didham (2008) note that forest beetle assemblages are affected by edge effects for up to 1 km from the edge, and many New Zealand forest birds are only found in heavily forested areas (Walker & Monks 2018).

While clearance of all vegetation types increased between 1989–2001 and 2001–2008, the vulnerability of native shrubland to loss has increased the most across all periods of the study, with forest vulnerability also showing a consistent upwards trajectory over time (Fig. 4l). Short-stature, native, non-woody vegetation, such as grasslands, went from second most likely vegetation type to be cleared in 1989–2001 to most secure by 2008–2015. We do not understand the drivers of the change in status of the different vegetation types, which could be due to changes in economic drivers, regulatory awareness, or just running out of accessible examples of the vegetation type. For instance, by 2008–2015 clearance of native non-woody vegetation was 519 ha (vs 577 for native shrubland), but much of the remaining 115 000 ha of native non-woody vegetation is in higher-elevation ecological districts, which underwent minimal clearance. Hence additional protection for lowland examples of native non-woody vegetation is still required.

Pressure to clear increasingly marginal lands (H3 and H8) was evident in several ways. First, there was a strong increase in the probability of clearance over time. Annualised probability of clearance more than doubled between 1989–2001 and 2001–2008, and clearance in 2008–2015 was still nearly double that seen in the early period of the study. Although the probabilities of loss measured in this study appear relatively small, they need to be considered in the context of the small amounts of indigenous vegetation and habitat left in New Zealand's most accessible and arable landscapes (Cieraad et al. 2015).

Second, between 1989–2001 and 2001–2008 the increase in development occurred largely on lower-angled surfaces, whereas by 2008–2015 clearance was across a much broader range of slope angles again (Fig. 4c). Production from steeper slopes is limited by accessibility and a tendency toward thinner soils and lower fertility, but dramatic increases in pasture production can be achieved through fertilising (Lambert et al. 2003).

Third, the increase in probability of clearance over time was mostly in drier areas with high Penman water deficits (Fig. 4d). Drier areas require more inputs (such as irrigation and fertiliser) to sustain moderate to high productivity because they are naturally less fertile and growth is seasonally limited by soil moisture (e.g. Brower et al. 2010; Monks et al. 2012). Interestingly, in 2001–2008 there was a weak trend for much of the increase in clearance probability taking place at sites with lower rainfall variability, whereas this effect had disappeared again by 2008–2015 (Fig. 4e). These data point to an initial increase in development in mesic areas, but in later periods this was less important, perhaps due to an increase in irrigation schemes. Irrigated land increased from around 260 000 ha to 460 000 ha across New Zealand between 1985 and 2002 (Parliamentary Commissioner for the Environment 2004), and nearly 800 000 ha of land is now irrigated, largely in Canterbury (64%), Otago (12%) and Marlborough (4%) (MfE & Stats NZ 2018).

Finally, increases in the probability of clearance over time were highest in the coldest areas (Fig. 4f). Temperature is a strong limiter of plant growth and soil mineralisation (Lambers et al. 1998), and so seasonal productivity and fertiliser inputs are key to the agricultural use of such lands. Part of this effect can be seen in the development of the inland South Island basins of Otago and Canterbury from 2001–2008 and from 2008–2015 (e.g. Spring Creek; Fig. 2).

While sector-based predictors were not the largest effects observed in the study, they do reflect some interesting patterns. Most clearance did not occur in areas with large amounts of horticulture and cropping (Fig. 4h). This pattern could be explained by little expansion in tilled land, as occurred in the period up until 2001 (MacLeod & Moller 2006). While viticulture increased dramatically over the period of the study (New Zealand Winegrowers 2010, 2017), many vineyards probably occupy areas of land where native cover had already gone (e.g. in Hawke's Bay, Marlborough and Central Otago). Similarly to cropping/horticulture, we found higher clearance in areas with little dairy for the first two periods of our study (Fig. 4g). The reverse pattern from 2008 to 2015 could reflect the maximisation of production of existing farms, or perhaps conversions of the remaining non-dairy farms within dairy-dominated areas. Land area under dairy increased by 42% from 2002 to 2016, from 1.8 million to 2.6 million ha (MfE & Stats NZ 2018).

The effects of surrounding exotic forestry area contrasted with those of cropping/horticulture and dairy. There was a higher probability of clearance in RAPs with more surrounding forestry from 1989 to 2008 and no relationship in 2008–2015. This pattern can be explained by two mutually compatible hypotheses. First, it is consistent with an expanding forestry industry through the 1990s and early 2000s until the price of carbon dropped in the mid- to late-2000s, precipitating a slump in new plantings (MfE & Stats NZ 2018). With the drop in new plantings there would have been less pressure to clear indigenous vegetation to facilitate exotic forestry. Second, our observed pattern may reflect an effect of Forestry Stewardship Council (FSC) certification, which increased after about 2000 and specifically prohibits the clearing of RAPs for new forestry plantings (Brockerhoff et al. 2008; Forest Stewardship Council 2013). However, this certification does not limit all indigenous vegetation clearance for forestry expansion because it is not universally adopted, especially by small operators, and we still recorded considerable clearance of RAPs from 2001 to 2008.

Our data cannot distinguish between the two explanations for the slowing of RAP clearance associated with forestry. However, we can resolve an earlier debate over the historical role of forestry in clearing indigenous vegetation and/or habitat. Walker et al. (2006) concluded that exotic forestry was one of the major causes of indigenous cover loss, but Brockerhoff et al. (2008) refuted this, identifying large errors in the identification of forestry areas in the LCDB1 and LCDB2 of the time. Crucially, Brockerhoff et al. (2008) were not able to present an unbiased assessment of forestry-driven change because they did not sample any areas that LCDB did not then identify as affected. Our models were designed to identify predictors of clearance rather than the actual cause of the change, and therefore only detected an association between nearby forestry and clearance. However, we can report that 35% of the areal change in RAPs from 1989–2001 was due to exotic afforestation and occurred largely in the northern Hawke's Bay and Gisborne areas. Some of this afforestation may have been on slips caused by Cyclone Bola. This figure dropped to 22% in 2001–2008, and to 5% by 2008–2015. All or most of the clearance in some ecological districts was due to afforestation (e.g. 95% of the 318 ha of RAPs cleared in Waiapu from 1989 to 2001; 96% of the 294 ha cleared in eastern Hawke's Bay from 2001 to 2008). Given that our assessment addresses only 'the best of what remained' on private land (i.e. RAPs) and not native vegetation generally, it will greatly underestimate forestry effects on indigenous vegetation and habitat. Hence it seems clear that, as concluded by Walker et al. (2006), exotic plantation forestry has been a major cause of loss of indigenous vegetation cover.

Despite identifying a number of factors associated with the clearance of RAPs, there is still considerable unexplained variance in the model (c. 60%). This uncertainty probably reflects the major roles that local decision-making, the values of landowners, and micro-topography play in the probability of RAP clearance. Many landowners probably value native biodiversity on their farms even if they do not seek formal protection for it. We have also not been able to incorporate the relative protection afforded by district plans and the interaction with vegetation type into the model. There is considerable variation between consenting authorities in terms of the types of vegetation that require consent to clear (Myers et al. 2013).

In order to gauge the impact of modern land management practices, we consider it useful to place the current risk to native vegetation in context by comparing it with historical patterns of anthropogenic deforestation. We have calculated comparable deforestation figures for the periods from human settlement (1280 AD; Wilmshurst et al. 2008) to 1840 (just prior to extensive European settlement), and from 1840 to 1970, both periods of major change in New Zealand land cover. We assume that at the time of human settlement, 80% of the land area of North, South and Stewart Islands was forested, and that this was reduced to 24% by the 1970s (Newsome 1987). McGlone (1989) estimated that nearly 50% of these forests had gone by the time of the first European surveys (1840–1860). Assuming 40% forest cover in 1840, an average of 0.12% of the remaining vegetation was cleared each year between 1280 and 1840. Between 1840 and 1970 the figure is 0.39% yr^{-1}. In the two most recent periods covered by our study, the observed clearance risk was 0.14% yr^{-1} (2001–2008) and 0.11% yr^{-1} (2008–2015). Contemporary risks of vegetation clearance in identified RAPs on private land were thus comparable with average risk of pre-European clearance, and approaching post-European clearance (the most intensive period of agricultural expansion in New Zealand history).

Conclusion

This is the first time the predictors of clearance of native-dominated fragments and/or native habitat on private land in agricultural landscapes have been identified. Our results paint a picture complementary to and consistent with other insights into the ongoing clearance of indigenous vegetation and habitats across New Zealand (Walker et al. 2006; Weeks et al. 2013; Cieraad et al. 2015). The progressive clearance of identified RAPs ('the best of what remains') over the last 26 years demonstrates that existing protections afforded through district planning processes are inadequate for maintaining ecologically significant native vegetation or habitats, and by extension native biodiversity, on private land. Recent rates of loss are comparable to deforestation in other periods since human settlement, and the natural protection afforded in the past by the low inherent productivity of marginal land has diminished as technologies and economic circumstances have changed. Formal legal protection has constrained clearance, but this protection is mainly provided for vegetation that private owners have no intention of clearing anyway. Thus the 'either/or' of production and conservation decried by Moller et al. (2008) has been reinforced.

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Supplementary material

Additional supporting information may be found in the supplementary material file for this article:

Table S1. Protected Natural Areas Programme survey reports used in this analysis.

Table S2. Reclassified vegetation classes and the LCDB4 classes merged to construct them.

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