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Subsistence lifestyles and insular forest loss in the Louisiade Archipelago of Papua New Guinea: an endemic hotspot

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Abstract. Insular areas of the south-west Pacific support high levels of global biodiversity and are undergoing rapid change. The Louisiade Archipelago of Papua New Guinea is a poorly known location with high levels of endemism. The largest island, Sudest Island, supports single-island endemic species and has the largest tract of forest remaining in this island group. The islands still support traditional subsistence lifestyles. This study investigated the patterns of forest loss since 1974 and predicted future forest loss to identify areas of conservation concern. We collected village population census data to assess population growth from 1979–2011. Historical vegetation mapping from 1974 was compared with Global Forest Change data from 2000–14. The geospatial drivers of forest loss were investigated using a generalised linear mixed model. Projected forest cover loss patterns in the islands were modelled in GEOMOD to the year 2030. Resident populations grew rapidly (6.0% per year, 1979–2011) but only a low rate of forest loss (e.g. −0.035% per year, Sudest Island) was observed between 1974 and 2014, restricted to low elevations near villages. Future modelling showed varied impacts on the remaining forest extents of the larger islands. The study offers a rare contemporary example of a biodiverse hotspot that has remained relatively secure. We concluded that local cultural and environmental settings of islands in the south-west Pacific can strongly determine the patterns and processes of forest cover change, and need to be considered in programs to conserve endemic diversity.

Additional keywords: biodiversity conservation, conservation biology, deforestation, island management, rainforest biodiversity, swidden

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Introduction

Anthropogenic habitat loss in native ecosystems is one of the greatest threats to global biodiversity (Brooks *et al.* 1997, 2002; Szabo *et al.* 2012). This is an increasing concern given the potential impacts of the growing human populations in developing countries, many of which are biodiversity hotspots (Cincotta *et al.* 2000). Tropical islands are abundant geographical features of the developing countries of the south-west Pacific. Many of these islands support unique endemism prone to habitat loss (e.g. Brooks *et al.* 1997, 1999; Szabo *et al.* 2012). However, the geographical context and unique evolutionary outcomes (e.g. Whittaker and Fernández-Palacios 2007) can also influence the rate of habitat loss and associated impacts on island biodiversity (Brooks *et al.* 1999; Ewers and Didham 2006). Understanding the mechanisms behind forest loss in understudied and biodiverse islands of the south-west Pacific remains a conservation priority, with rapid environmental changes occurring in the region (e.g. Shearman *et al.* 2009; Miettinen *et al.* 2011; Margono *et al.* 2014).

Agricultural expansion is the main land-use associated with global tropical forest loss (e.g. 96%: Geist and Lambin 2002). In tropical developing countries, this is primarily through commercial agriculture, followed by subsistence agriculture (Hosonuma *et al.* 2012; Houghton 2012). Subsistence agriculture, in the form of traditional shifting cultivation (swidden) practices, has remained a core activity for both cultural reasons and to maintain livelihoods in many developing nations (Lambin *et al.* 2001; van Vliet *et al.* 2012; Li *et al.* 2014). However, controversy surrounds the impacts of this practise on biodiversity relative to more intensive alternatives (de Jong 1997; Padoch and Pinedo-Vasquez 2010; Ziegler *et al.* 2011). In general, subsistence agriculture is considered an early stage of the land-use transition process, which with time will likely shift to more intensive land uses (Foley *et al.* 2005). This transition away from subsistence agriculture is evident from numerous regions of the tropics (e.g. Fox *et al.* 2009; Mertz *et al.* 2009; van Vliet *et al.* 2012). Though impacts on biodiversity are often strongly linked to population density and poverty levels, local-scale factors are a key consideration in understanding subsistence agriculture, and its impacts on biodiversity (see Geist and Lambin 2002; Carr 2004; van Vliet *et al.* 2012). These factors include sociocultural context, government/conservation policies, market access and interactions with regional or global forces (Lambin *et al.* 2001; Geist and Lambin 2002; Fox *et al.* 2009).

The island of New Guinea contains the third-largest tract of tropical rainforest remaining globally, and its eastern half, Papua New Guinea, supports diverse and unique ecosystems that are globally significant (Mittermeier *et al.* 1998; Myers *et al.* 2000; Brooks *et al.* 2006). Several Papua New Guinea endemic hotspots have been identified as conservation priorities, many of which have human populations heavily reliant on subsistence agriculture. These include insular areas such as parts of the East Melanesian Islands Hotspot (Critical Ecosystem Partnership Fund 2012) and the Louisiade Archipelago, south-east Papua New Guinea (Endemic Bird Area: BirdLife International 2013). However, context-specific data useful to conservation efforts remain largely lacking for these remote islands, particularly for the Louisiade Archipelago. To date, no studies have specifically investigated mechanisms driving forest cover change in the Louisiade Archipelago. Of concern is the large logging concessions that predict future forest losses on Papua New Guinea's islands (Shearman and Bryan 2011). Consequently, pressures upon the forests that support endemic biodiversity in the Louisiade islands are likely to increase into the future (Shearman and Bryan 2011; Goulding *et al.* 2016). Commercial logging and plantation operations are largely absent but remain a threat with increasing global demand for resources such as tropical timbers, gold and palm oil (Koh and Wilcove 2008; Shearman *et al.* 2012; Alvarez-Berríos and Aide 2015). The most geographically remote eastern islands (Sudest and Rossel) of the Louisiade Archipelago are locally renowned for strong cultural beliefs and practises (e.g. Lepowsky 1990, 1991; Shaw 2016). Of these, Sudest Island has the largest tract of forest supporting unique endemism across a range of taxa (e.g. Slapcinsky 2006; Johns *et al.* 2009; Kraus 2009) and, of note, endemic bird species representing close to 5% of the global Data Deficient bird species (IUCN 2016). Understanding patterns and drivers of forest cover change in the absence of large-scale commercial land-use activities on these islands remains a priority.

Here, we investigate the mechanisms of forest loss within the local cultural context of the Louisiade Archipelago. We focus on Sudest Island, revealing the environmental and cultural intricacies requiring consideration in future biodiversity conservation efforts on similar islands of the south-west Pacific. Our specific aims were to (1) quantify the recent rates and patterns of forest cover change since 1974, (2) investigate the mechanisms driving this change, and (3) predict future forest cover change in areas of conservation concern.

Methods

Study area

We restricted the study to islands of the East Calvados Chain and the largest forested island in the Louisiade Archipelago, Sudest Island (11.5°S, 153.5°E) (Fig. 1). Islands of the East Calvados Chain were included because they are biogeographically similar to Sudest Island and support important subsets of the endemic forest diversity, including some of the endemic Data Deficient bird species.

Demography

Human population data were accessed from local government census data for the years 1979, 1999 and 2011 (local government sources, 2013). We extracted total population counts from these data and measured the percentage annual population growth rates for different islands over different census periods. Difficulties in the accuracy at the village-level census data limited the use of geospatial population data within the local government ward. Average population growth rates for islands and wards were calculated as follows:

$$\%Annual\ Growth = \frac{\frac{Later\ pop. - Earlier\ pop.}{Earlier\ pop.} \times 100}{Time\ frame\ in\ years}$$

Subsistence gardens

We accessed Google Earth imagery (8 August 2006, 25 December 2015) to randomly select and measure the size of subsistence gardens. Deforested areas with regular boundaries were interpreted to be subsistence gardens rather than for another land use. This assumption was based on ground observations of the study islands.

Forest loss 1974–2000 – Sudest and Junet Islands

We mapped vegetation classes derived from aerial photography of Sudest Island by the Royal Australian Survey Corps (1973–75) to determine the approximate forest extent (to ±25 m accuracy) in 1974. These vegetation classes were digitised, georectified in ArcGIS, projected, resampled and mosaicked to match Global Forest Change data (Hansen *et al.* 2013) then classified into two classes: forest and non-forest. The resolution and accuracy of the historical aerial photography were adequate to detect any drastic land-cover changes within this period. Similar techniques have been used effectively to estimate land use/land cover changes elsewhere in the region (e.g. Ningal *et al.* 2008; Shearman *et al.* 2009).

The 1974 mapping data were then compared with the Global Forest Change data for the year 2000. This forest dataset classifies forest as any vegetation over 5 m high (Hansen *et al.* 2013), reducing the ability to detect and compare changes with different stages of succession. The thresholds in the percentage canopy closure between undisturbed forest and non-forest categories may be context specific, and vary between on-the-ground measures and forest-cover products (GFOI 2016). We conducted ground-truthing in disturbed and undisturbed interior forests to identify a minimum forest-cover value representative of relatively undisturbed forest. We then conservatively defined forest as cover values >75%.

Forest loss 2000–14 – all islands

Spatial data preparation was conducted using the software ArcGIS 10.1 (Esri, CA, USA). Annual forest disturbance was defined as the annual amount of forest lost (stand replacement) between 2000 and 2014 (Hansen *et al.* 2013). We did not consider Global Forest Change forest gain data in this analysis. Caution is advised when comparing data between 2000–12 and 2011–14 because of the inclusion of Landsat 8 OLI (Operational

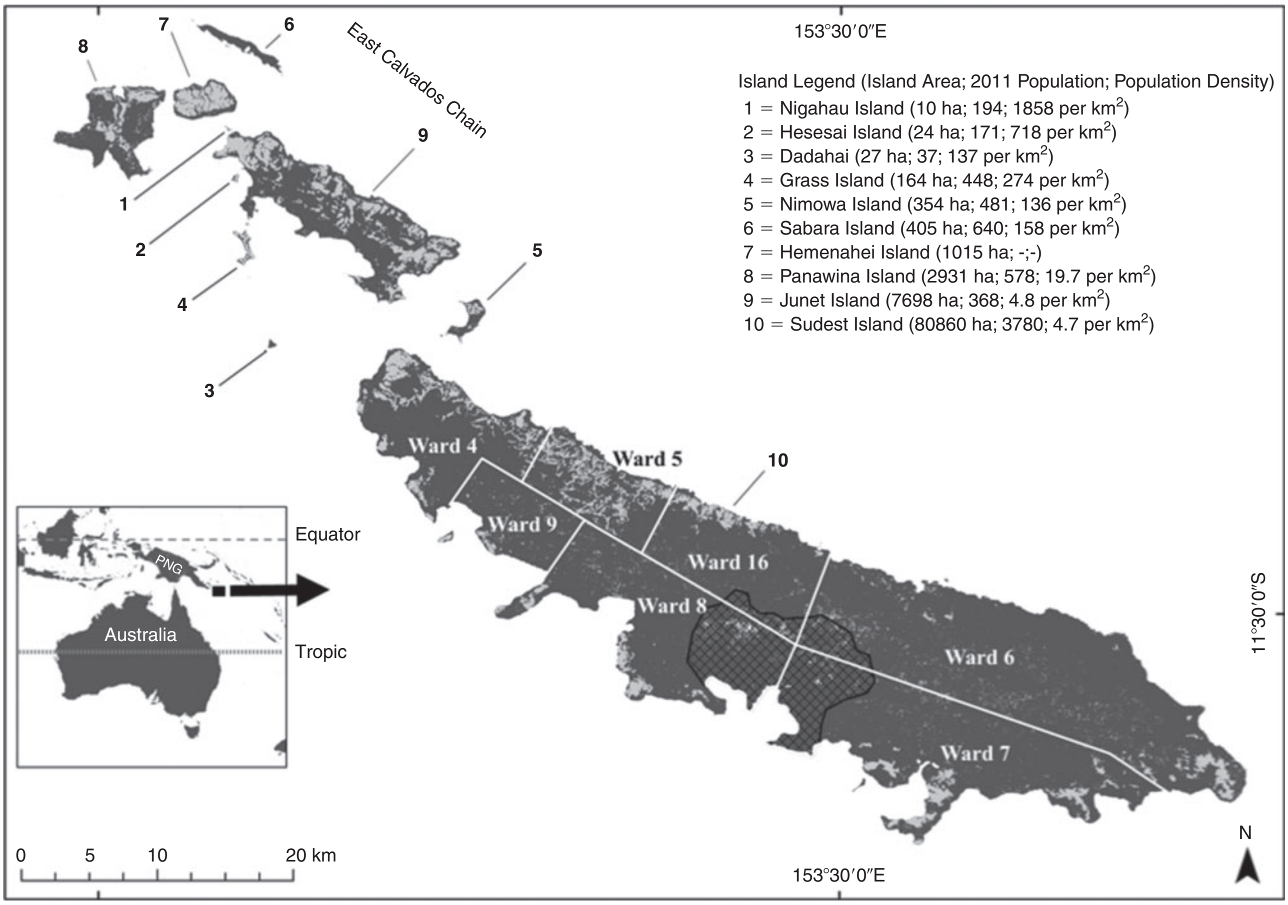

Fig. 1. Location of the study islands in the Louisiade Archipelago, Papua New Guinea. Local government ward boundaries are delineated on Sudest Island. Forest areas are represented by dark shading, non-forest areas by light shading. Mapping is based upon Global Forest Change data for 2000 (Hansen *et al.* 2013). Local resident population data (from the 2011 local government census) and population densities follow island area in the figure legend. The cross-hatched area delineates the culturally controlled area, Bwamumu.

Land Imager) sensor that may change detection levels (Hansen *et al.* 2013). We used Getis–Ord GI* spatial analyses to test for spatial autocorrelation in the land-cover data and to identify spatial clustering in forest loss between 2000 and 2014 (Getis and Ord 1992).

The percentage of annual forest loss (r) was calculated following Puyravaud (2003) as:

$$r = \frac{1}{t2 - t1} \ln \frac{A2}{A1}$$

where $t2$ and $t1$ are the later and earlier years (respectively) and $A2$ and $A1$ are the later and earlier forest areas, respectively.

We modelled predictor variables to understand the current drivers of change and to inform driver choice for a model to predict future forest loss.

Modelling predictors of forest cover change 2000–14

We used a mixed-effects logistic regression model in the lme4 package (ver. 1.1–13) (Bates *et al.* 2015) with island as the random effect to analyse patterns of forest cover change for 2000–14. All analyses were conducted in R 3.3.2 (R Core Team 2016). The fixed effects were elevation, slope, distance to the nearest village and preclearing forest-cover values. These predictors were chosen on the basis of field observations and the literature. We used Aster Global Digital Elevation Model 2 (2011) data for elevation. Preclearing forest-cover values were extracted for areas of forest loss and non-loss. Village locations were extracted from topographic maps (Royal Australian Survey Corps 1973–75) and ground-truthing during field surveys. We used proximity tools and the slope function (Spatial Analyst Tools; ArcGIS) to calculate the remaining variables (Distance to village; Slope). Data were aggregated ($\times 8$) due to the large size of the dataset ($>$1.2 million data-points), while considering capture of the geospatial patterns of forest-cover change within the limits of analyses and a GLM. This resulted in ~23 000 data points, representing areas both disturbed and undisturbed.

All predictor variable values were standardised to allow direct comparison of parameter estimates. Data were visually inspected with plot functions for collinearity. A colinear relationship ($r = 0.682$, $P < 0.01$) was found between slope and elevation. Univariate models were run for each predictor variable and for the null model. Elevation had higher predictive power than slope, and so the latter was discarded from further analyses. We selected a subset of variables for multivariate modelling. Model residuals were tested using the ncf package to check for spatial autocorrelation (ver. 1.1–7) (Bjornstad 2016). An autocovariate was introduced for model improvement given the likelihood of the effects of spatial autocorrelation (Crase *et al.* 2012). We then used the dredge function in the MuMIn package to assess model performance (Bartoń 2016).

To investigate whether forest locations further from villages were being disturbed with increasing time we used an Analysis of Variance (ANOVA) to test for the effect of year on the distance of new gardens from villages (predictor). A Bonferroni *post hoc* test was used to further investigate changes in relationships between 2000 and 2012. We only included these years because of the potential issues with altered detection of forest change following the use of Landsat 8 OLI in 2013 and 2014 (Hansen *et al.* 2013).

Predicting future forest loss

We used the predictive landscape change analysis modelling tool GEOMOD (Pontius *et al.* 2001) in TerrSet software (Clark Labs 2015) to predict future forest loss. This model is suited to the binary nature of the Hansen *et al.* (2013) forest-loss dataset. Distance to the nearest village and elevation, which had high predictive scores of forest loss between 2000 and 2014, were selected from the mixed-effect modelling selection process. We tested different weightings of these drivers in model runs but found equal weightings offered the best outcomes.

We used the classified Hansen *et al.* (2013) land-cover image for the year 2000 as a starting raster to generate a predicted 2007 forest-cover image using the aforementioned drivers. The actual 2007 forest-cover image was constructed by removing the observed forest loss for each year between 2000 and 2007 from the original 2000 land-cover image. We used the 2007 forest-cover data to validate the modelled forest-cover change. The same process was repeated for time steps between 2007 and 2014, and 2000 and 2014. We used the observed forest loss between 2007 and 2014 to predict forest loss to 2030. Pixels lost between these periods were summed and used to project annual forest loss (r) (Puyravaud 2003) until the year 2030; that year was chosen because the time frame was similar to the 2000–14 timespan, for which we could inspect model accuracy using K_{no} and $K_{location}$ values.

Results

Population growth

The local resident population in the study islands increased from 3579 in 1979 to 5632 in 1999 (2.9% per year) reaching 10 477 by 2011 (7.2% per year, 1999–2011; 6.0% per year, 1979–2011). At a finer scale, patterns in human population growth rates showed greater variation across islands and between census periods. Population densities were highest on the smallest islands, which in some census periods experienced negative population growth rates. In contrast, the largest two islands supported lower densities of ~4.7 persons per km^2. Sudest Island's north-western wards (Wards 4, 5 and 9) supported the highest human population densities and the highest population growth rates during the latter census period of 1999 to 2011. Overall, Nimowa Island had the highest population growth rate over the entire census period (~9.0% per year), followed by Panawina, Grass, Junet and Sudest Islands respectively (~3.9 to 2.3% per year) (Table S1, available as Supplementary Material to this paper).

Garden expansion

Garden sizes showed a large variation in both their location and size within islands, ranging from 0.02 to 0.75 ha with a mean of 0.15 ± 0.12 (s.d.) ha ($n = 133$). Elevation of deforested areas ranged from 0 to 669 m (above sea level), with an average of 76 ± 75 (s.d.) m (above sea level) ($n = 4535$). More than 99% of forest loss was below 236 m above sea level. The mean distance from villages at which forest loss occurred was 2273 ± 1972 (s.d.) m ($n = 4535$).

Forest loss 2000–14 – all islands

There was a general trend of the smaller islands supporting the least forest cover and displaying the most variation in annual forest loss during the 2000–14 period (Fig. 2). Low island proportions of forest-cover change occurred on the intermediate-sized islands Sabara and Hemenahei, and the three larger islands (Panawina, Junet, Sudest). Panawina Island had scattered forest loss in its southern and eastern forest extents. However, the largest hotspots of forest loss were concentrated on the two largest islands (Junet and Sudest) during the same period (Fig. 3). There was a decrease in the forest cover on Junet Island of 0.24% per year, which was double the amount observed on Sudest Island (0.12% per year). Forest loss since 2000 has predominantly occurred on the western side of the island, particularly in a large south-western patch closer to Grass and Hesesai Islands (Fig. 3).

Forest loss 1974–2000 – Sudest and Junet Islands

We observed little forest cover change between 1974 and 2000 on Sudest Island (Fig. 4). Areas classed as secondary growth and plantations in 1974 covered ~1.3% of the island area. Grasslands/non-forest areas covered ~6.2% with forest the remainder (92.5%). Over the 25 years, there was a 0.85% of island area reduction in forest cover or a −0.04% per year decrease in forest cover. Some areas that were classified as non-forest in 1974 had regenerated to a forested state by 2000. In other areas, non-forest encroached into forest extent during the same period. Wards 4, 5 and 9 had the highest rates of forest loss on Sudest Island between 2000 and 2014 (Table S1). The second-largest extent of forest occurred on Junet Island. Greater forest-cover change was observed on this island between 1974 and 2000 than on Sudest Island. In 1974, secondary growth and plantations covered 0.9% of the island. Grassland/non-forest areas covered 16.9% of the island and forest covered the remaining 82.2%. Forest extent had reduced by more than 10% to cover only 71% of the island by 2000 (−0.44% per year) (Fig. 4).

The ANOVA for distance from village between years found year had a significant effect on distances ($F_{11,5456} = 30.251$, $P \leq 0.0001$). The *post hoc* test revealed that in 2005 and 2010/11 there were increased distances in deforestation from villages

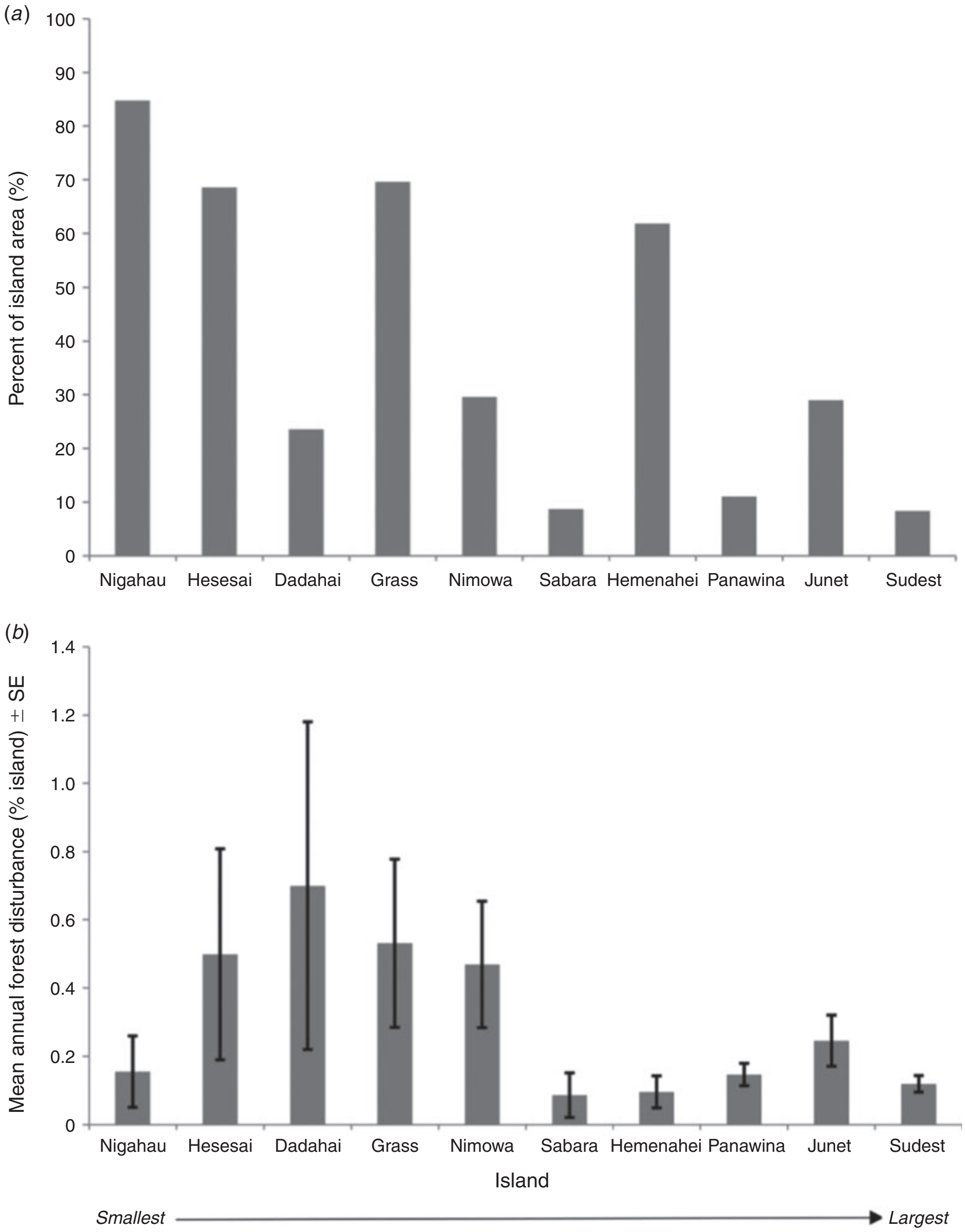

Fig. 2. (*a*) Proportion of island area with less than 75% canopy cover in the year 2000, and (*b*) mean annual percentage of total island area disturbed between 2000 and 2014. Islands are ranked in order of size from left to right.

(Fig. 5). Other years showed more similarities amongst them, with the overall pattern appearing cyclical.

Results of the binomial logistic mixed-model supported autocorrelation effects (e.g. see Fig. 3; Fig. S2, Supplementary Material) with model improvement with the introduction of an autocovariate in the analyses (Table 1). Slope consistently had weaker predictive estimates than elevation in all test model runs, both as an individual driver and in the complete interaction best model. Distance to village, on its own, was a weak predictor of forest loss but had a stronger interactive effect with both elevation and forest cover (Table 1).

Future model prediction

The GEOMOD land-change model prediction for the period from 2014 to 2030 demonstrated greater forest loss than for the 2000–14 period (~550 ha) (Table 2). This follows the observed trend of increasing forest loss over the 2000–07 and 2007–14 periods. This amounted to a 50% increase in forest loss in 2007–14 when compared with 2000–07. Overall, the model prediction for 2030 showed a consolidation of forest loss close to previously disturbed areas, and only relatively minor expansion of forest loss into new areas (see Fig. 4). The model accuracy for location remained high,

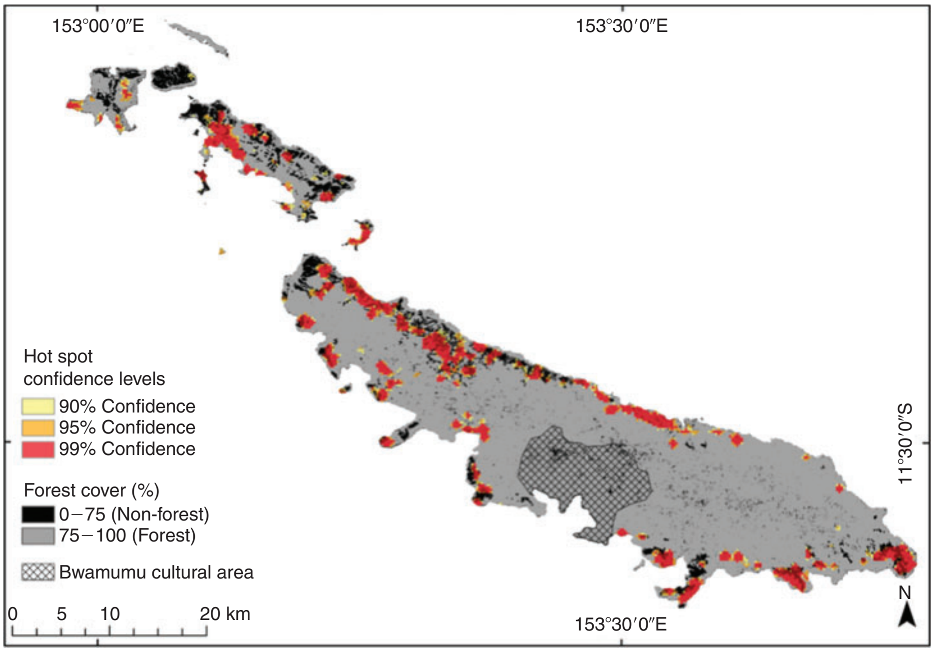

Fig. 3. Hot spot results of a Getis–Ord GI* analysis for spatial clustering of forest loss between the years 2000 and 2014. To aid in visualisation of the hot spots, forest areas are shaded light grey and cleared areas as in black. Forest extent data are from Global Forest Change data for the year 2000 (Hansen *et al.* 2013). Bwamumu, an important cultural area where most activities are prohibited, is delineated by hatching.

with the lowest $K_{location}$ value being 85% agreement over the 2000–14 period.

Discussion

Main findings

The smaller study islands supported the highest density of local residents and exhibited the highest non-forest proportions and variation in mean annual forest disturbance. Population density and access to close-proximity low-elevation areas of forest were significant determinants of the spatial patterns in forest loss in the Louisiade Archipelago. Overall, the observed spatial and temporal variation in forest-cover change was driven by factors at the community level and in response to local socio-environmental conditions across adjoining islands. The largest concern remains the continuing erosion of important remaining lowland forest areas supporting endemic diversity on parts of Sudest Island, and, in particular, on Junet Island. Large areas of Sudest Island, either those culturally protected, at higher elevations or distant from villages, have shown little change historically or with model prediction into the future. This is because forest loss in these islands has remained largely independent of exogenous forces and has proceeded at a relatively slow rate compared with elsewhere in Papua New Guinea (e.g. Buchanan *et al.* 2008; Shearman *et al.* 2009; Shearman and Bryan 2011). This offers a rare contemporary example of a biodiverse hotspot that has remained relatively secure. This could be expected to continue only in the unlikely event (e.g. Laurance 2007; Lambin and Meyfroidt 2010) that endogenous practises and mechanisms driving forest-cover loss persist unchanged in these islands (although see Mertz *et al.* 2012), and if exogenous pressures such as commercial logging and plantations remain absent.

Patterns of forest cover change

Papua New Guinea had a national annual population growth rate of 2.48% during the period of 1979 through to 2015 (World Bank 2017). This is much higher than the global average for the same period (1.47%), and second only to the Solomon Islands in Melanesia (2.69%) (World Bank 2017). Collectively, the local populations of the study islands increased at more than double the national rate largely due to growing populations on Nimowa, Grass and Panawina Islands (Table S1).

The size of areas cleared for subsistence gardens in the study islands (mean 0.15 ha) are at the small end of the observed scale for subsistence gardening. However, they are comparable to those of other rainforest areas in Melanesia that are generally well under 1 ha (Kinch 1999; Umezaki *et al.* 2000; Mertz *et al.* 2012) and both smaller than, or similar to, the areas cleared for this purpose in other tropical areas globally (e.g. Aweto 2013). Similar to other findings for Papua New Guinea (Shearman *et al.* 2009), the pattern of forest loss observed on these islands is strongly tied to population density. This is an obvious pattern that is in contrast to other areas worldwide that are increasingly subject to greater interaction with external forces (Lambin *et al.* 2001). A second major determinant of the spatial patterns observed relates to the availability of low-elevation land close to villages. The largest human populations are on the smallest islands, which in turn are the most restricted with respect to suitable land available for gardening. This is reflected in the low forest cover and sporadic clearing observed on the seven smaller

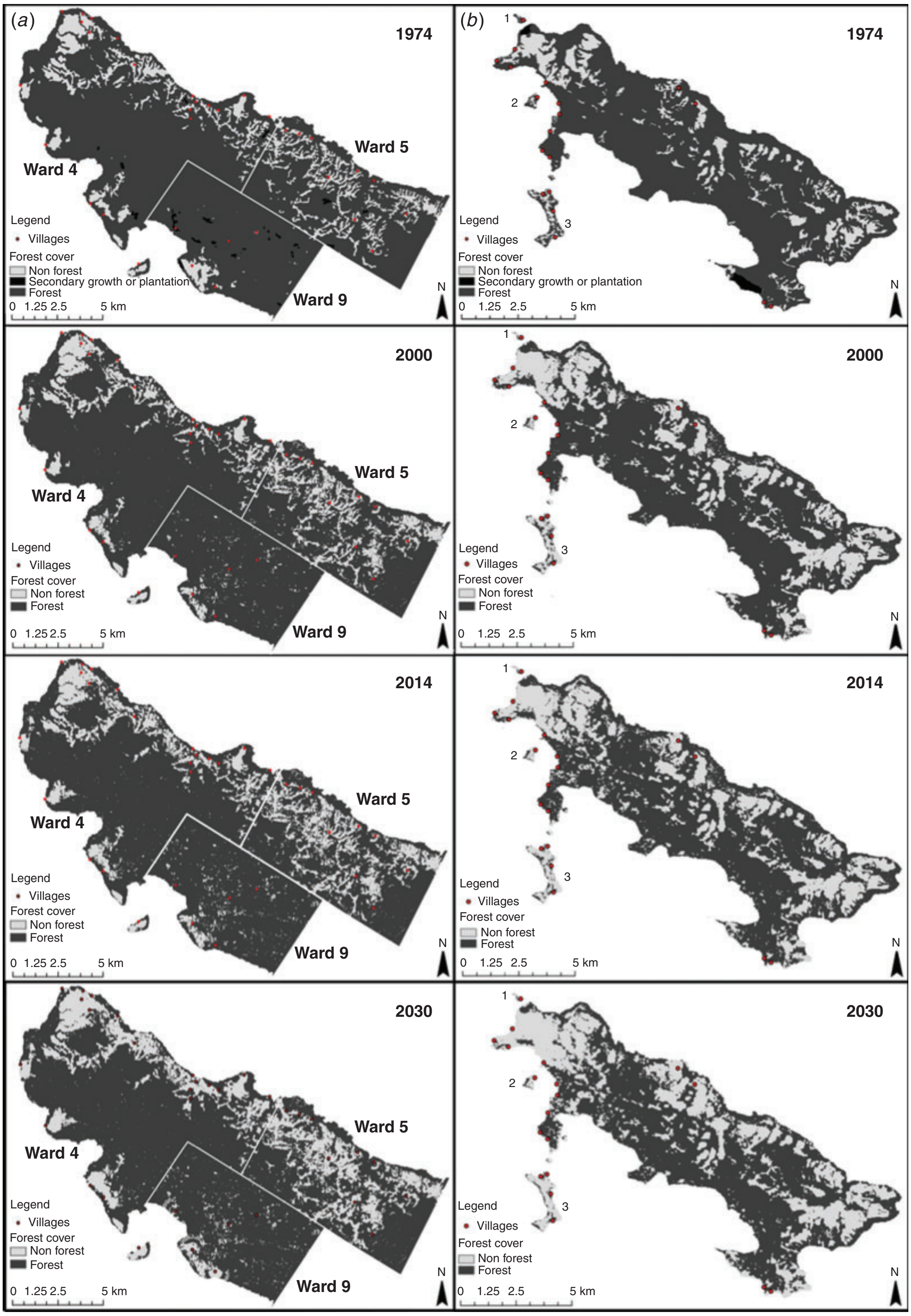

Fig. 4. Historical and projected forest cover loss in the three wards experiencing the highest rates of forest loss on Sudest Island (*a*) and on Junet Island (*b*), which supports the second largest area of forest extent in the study islands. The smaller Nigahau, Hesesai and Grass Islands are also shown (1–3, respectively). Data for the year 1974 includes secondary regrowth and plantations (Royal Australian Survey Corps 1973–75). These categories are not defined in the later (Hansen *et al.* 2013) data or the modelled projection for 2030.

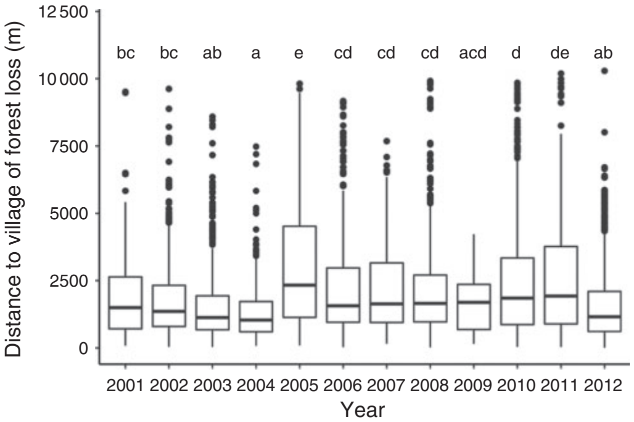

Fig. 5. Boxplot of raw data for the distances of deforestation from villages for each year. *Post hoc* results are displayed on top, where similarities are indicated through shared letters. As an example, 2005 is uniquely labelled with the letter 'e', which 2011 also features; whereas, 2001 and 2002 display similar distances (bc), as do 2003 and 2012 (ab).

islands (Fig. S1, Supplementary Material). Consequently, large portions of these populations concentrate their gardening activities on nearby Junet Island. This is most obvious amongst Sabara islanders, who reside on an uplifted coralline island with poor soil development and a lack of fresh water. This is also reflected by the negligible forest loss and forest perpetuation on this island. These residents undertake daily trips to nearby Junet and, to a lesser extent, Panawina, to conduct gardening activities and collect fresh water. However, the local residents of Grass Island, Nigahau, Hesesai and some Nimowa residents also follow a similar strategy on Junet Island (see also Hide *et al.* 2002). Consequently, understanding the heightened pattern of forest loss on Junet Island entails consideration of the local residents of adjoining islands.

Road infrastructure is known to be an important driver of the spatial pattern of forest loss and subsistence agriculture elsewhere (Allen and Barnes 1985; Lambin *et al.* 2001; Laurance *et al.* 2009). However, the lack of roads on these islands combined with coastally located villages whose inhabitants are dependent on either canoes or traversing by foot, restricts gardening activities to easily accessed, low elevations. This means that a key driver of forest loss patterns in these islands remains village location. Whether this changes in the future is yet to be seen.

A level of caution is needed when comparing rates of forest loss due to the different methods used (Puyravaud 2003). Conservatively, Papua New Guinea's forests are estimated to have declined around −0.47 to −0.49% per year between 1990 and 2010 (FAO 2010). Other estimates have placed this rate much higher, especially when including commercial activities, for example, reaching peaks higher than −1.5% nationally in the 1990s (Shearman *et al.* 2009). In this context, the focal islands in this study have relatively low annual rates of forest loss, particularly when compared with lowland forests elsewhere in Papua New Guinea. This is evident through the degradation or loss of ~30% of Papua New Guinea's lowland forests between 1972 and 2002 (Shearman and Bryan 2011) or the rapid loss

Table 1. Results of model selection using the dredge function (MuMIn package: Bartoń 2016) for predictor variables and forest loss

AC, autocovariate; E, elevation; FC, forest cover; DV, distance to nearest village. See Table S2, Supplementary Material, for full results

Model	Intercept	AC	E	FC	DV	E : FC	E : DV	FC : DV	E : FC : DV	d.f.	AIC	ΔAIC	Rank
1	−1.399	1.127	0.197	−0.317	0.004	−0.431	−0.081	0.102	0.229	10	17331.884	0	1
2	−1.479	1.169	0.145	−0.177	0.059	−0.279	−0.118	–	–	8	17436.438	104.554	2
3	−1.477	1.169	0.144	−0.181	0.060	−0.277	−0.118	−0.007	–	9	17438.368	106.484	3
4	−1.481	1.141	0.097	−0.127	0.047	−0.234	–	–	–	7	17470.227	138.343	4
5	−1.482	1.117	0.106	−0.130	–	−0.297	–	–	–	6	17471.112	139.228	5
AC	−1.683	1.120	–	–	–	–	–	–	–	3	17574.053	242.169	18
Null	−0.710	–	–	–	–	–	–	–	–	2	21305.259	3973.375	38

Table 2. Change periods used in the prediction of future loss of forest-cover, showing mean rate of observed forest loss (*r*), the area lost and Kappa number (agreement in terms of the quantity of cells in each category) and location (agreement in terms of the location of cells in each category) values for model accuracy. The Kappa values represent the accuracy (0–1) in predicted versus realised forest loss pixels in number and location (respectively)

Change period	Mean r	Total loss (ha)	Model K_{no}	Model $K_{location}$
2000–07	−0.007	658.4	0.973	0.936
2007–14	−0.004	992.4	0.960	0.912
2000–14	−0.006	1650.8	0.935	0.858
2014–30	−0.004	2200.5	–	–

(~20%) of lowland forests on the island of New Britain within a decade (Buchanan *et al.* 2008). The low forest-cover change is perplexing given the continuing population growth in the study islands and is likely to change should outside forces grow in prominence. We expect this might reflect the unaccounted for secondary forest regrowth and fallow areas that are generally reused. However, our predictions indicate little change in the near future (2030) if the current mechanisms driving forest-cover change continue.

Process

Inhabitants of islands in the Louisiade Archipelago are part of the Kula trade ring, where customary exchange amongst islands, particularly associated with mortuary feasts, plays a continuing and important role (Lepowsky 1991). The regular interactions and cultural contiguity amongst these islands allows the adoption of strategies beyond the island of residence (Hide *et al.* 2002). This reflects an important overcoming (temporary reprieve) of the interactions between increased local human population densities and reduced immediate land availability, i.e. those that drive greater environmental impacts with a reduction in fallow periods and more intensive land use (Umezaki *et al.* 2000; van Vliet *et al.* 2012; Aweto 2013). However, this strategy does not result in a reduction in the ongoing environmental pressure on the smaller densely populated islands, where fallow periods on easily accessible land remain shorter (cf. Sudest Island: Hide *et al.* 2002) and environmental disturbance intense (e.g. Fig. 2*b*).

The key finding of the importance of distance to village in forest loss seems obvious but also provides insight to historical impacts. This is not unusual (e.g. Mon *et al.* 2012) but highlights the importance of village locations and factors affecting these. Most villages in these islands are coastally situated. However, this was not always the case. Prior to European arrival and influence, warring parties from rival islands were common, with villages situated inland for greater protection (Lepowsky 1991). Local residents remain aware of many of these locations on Sudest Island, which often still bear signs such as betelnut palms (*Areca catechu*). This implies that even inland forests on the larger islands have a history of disturbance from subsistence gardens, and raises the issue of the level of pristineness of the forests (Willis *et al.* 2004). Similar examples of such changes exist elsewhere in Melanesia (e.g. Bayliss-Smith *et al.* 2003).

The ongoing belief in witchcraft in the eastern Calvados Chain and sorcery on Sudest Island plays a role in spatial patterns of subsistence gardening. In some cases this can drive people to move and form new hamlets in new locations or avoid others such as Bwamumu on the southern side of Sudest Island (Lepowsky 1990). In part, this may have had a role in determining smaller islands as the locations for facilities such as schools and hospitals (leading to larger populations), for example on Nimowa and Grass Islands. Of note, the most easterly Rossel Island (not included in this study) is renowned and locally feared amongst many of the inhabitants of these islands for strong cultural values, sacred places and witchcraft (Lepowsky 1990). Such reasons are often cited by inhabitants of mainland Milne Bay Province or areas outside the Louisiade Archipelago for not wanting to go there. In other words, these beliefs combined with the strong customary land ownership typical in Papua New Guinea, dissuade any form of transmigration that has been associated with forest loss in other countries (Fearnside 1997; Geist and Lambin 2002).

The processes driving subsistence gardening in these islands have also changed over the temporal scale. The use of forest resources and sago palm (*Metroxylon sagu*) was greater in the past with less emphasis placed on subsistence gardening (Lepowsky 1991; Kinch 1999; Hide *et al.* 2002). Garden areas are thought to have become bigger with the encouragement of the national government, and the introduction of metal tools and novel food plants (Lepowsky 1991). However, the practice of subsistence gardening is for more than just sustenance, it is a cultural practise that has remained strong with cultural ties to endemic species. As an example, local residents value and remain attuned to the vocal behaviour of the endemic Tagula butcherbird (*Cracticus louisiadensis*) because it acts as a prompt for gardening activities. The perceived value attributed to such species would be expected to last only as long as the cultural beliefs behind them. Another notable cultural aspect relating to gardens is the relationship with mortuary feasts (Lepowsky 1991). Feasts are an important part of trade relationships and indeed are expected when deaths occur. Planning can take considerable time, with families of the deceased cutting forest and creating new gardens well in advance to be able to provide enough food for guests at the mortuary feast. In some cases, there may be multiple feasts for an individual, increasing the need for new gardens. This demonstrates an unusual association with health as a driver of forest disturbance in the islands.

Regional and global climate patterns are known to have impacts on both the forests and gardening practices in the region (Johns 1986, 1989; Kinch 1999; Goulding *et al.* 2016). Severe droughts can restrict gardening practices and cyclones can destroy existing gardens and promote the formation of new gardens. We do not know what happened in 2005 and, to a lesser extent, 2011 that caused an increase in deforestation more distant from villages. A lack of accurate local weather data limits us to surmise that these were during drought periods, given that 2005 also had low overall forest loss. However, it is El Niño events in the south-west Pacific that bring drier than normal conditions. Historical weather records of the El Niño Southern Oscillation/El Niño (ENSO) events do not support this, with the period around 2005 being a neutral ENSO period and 2011 during a strong wet La Niña phase (Australian

Government Bureau of Meteorology 2017). Similarly, cyclones can cause changed practices or widespread (distant from villages) forest damage that could appear as anthropogenic deforestation (Goulding *et al.* 2016), but none passed through the Louisiade Archipelago in these years. One possible explanation is that 2005 was also a particularly good year for Bêche-de-Mer harvesting, leading to less reliance on gardening (Fr A. Young, pers. comm.). There was a peak in national exports of this marine product observed during 2005/06, before the moratorium on harvesting came into place in 2009 (Kinch *et al.* 2008). If this were the case, it would be an interesting example of the connectivity of anthropogenic threats across marine and terrestrial realms, a subject highly relevant to conservation efforts (e.g. Beger *et al.* 2010).

Unlike forest loss, forest gain is a gradual process that may take 15–30 years or more to reach a secondary successional stage in rainforest that supports sufficient functional diversity (Whitfeld *et al.* 2014). Complete forest regeneration is likely to require well beyond 100 years to reach undisturbed mature forest diversity values, depending on the location and disturbance frequency (Finegan 1996; Cole *et al.* 2014). Consequently, the continuing albeit slow forest disturbance observed across these islands is likely to have had long-lasting effects on the forest structure and species composition (e.g. Johns 1986). Support for this comes from the fact that anthropogenic activities have influenced forest composition even in the largest remaining tract of rainforest globally, in the Amazon basin (Levis *et al.* 2017). Furthermore, the ecological impacts of anthropogenic activities are generally greater on islands than mainland areas (Cronk 1997) with even minor forest disturbance capable of impacting specialised endemic island species (Trainor 2002, 2007; Davies *et al.* 2015). How future changes will affect the endemic forest diversity of these islands is unknown. What we do not know, but suspect, is that many of the endemic species that occur on these islands today are the product of coping with this human occupation history, exhibiting a certain level of resilience in some of these disturbed landscapes.

Approach and limitations

The long-term analysis of forest-cover change extending back to 1974 improved insight into the spatial patterns and rates of forest loss in these islands. Our analyses were limited through not measuring secondary growth, plantations and forest gain during the study period, 2000–14. As a result, the findings better reflect forest disturbance during this time-frame, an important determinant for many endemic island species. We expect the lack of differentiation of these other vegetation classes could result in an overestimation of primary forest cover at low elevations and on the smaller islands. The findings would be improved with data over a longer period, particularly in relation to better understanding the relationship of forest-cover change in these islands with predicted climate change and regional weather patterns, such as El Niño/La Niña events. Regardless, this study does show that valuable conservation insights relating to forest-cover change can be drawn from combining global remote-sensing datasets with localised knowledge of important island hotspots of endemicity.

Conclusion

The study islands of the Louisiade Archipelago are under increasing population pressures but their forests appear to show little change over the study period and projected into the future. The current drivers of change are determined by isolation and the continuing strong cultural context. This remains a relatively positive situation for the endemic bird diversity, especially on Sudest Island, the largest island of the archipelago. However, important subsets of these species harboured on adjoining islands are likely to be heavily impacted into the future, particularly those reliant on the second-largest extent of forest on Junet Island. Furthermore, should external forest disturbance pressures, such as logging and plantations, increase in the future or the current cultural values change (Fazey *et al.* 2011), then the remaining forested areas of Sudest Island are likely to become under pressure. Given the difficulties in meeting their needs, solutions that can help local residents cope with change while maintaining their strong cultural beliefs would be the best option for forest conservation in these islands.

Conflicts of interest

The authors declare no conflicts of interest.

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Pacific Conservation Biology

Supplementary material for

Subsistence lifestyles and insular forest loss in the Louisiade Archipelago of Papua New Guinea: an endemic hotspot

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Table S1. Islands ranked in increasing size with associated demographic data (1979-2011) and forest loss rates (2000-2014).

Island	Area (ha)	2011 Population	2011 Pop. Density (km^2)	Pop. Growth Rate '79-'99 (%/yr)	Pop. Growth Rate '99-2011 (%/yr)	Total Pop. Growth Rate (%/yr)	Forest Loss 2000-2007 (*r*)	Forest Loss 2007-2014 (*r*)	Forest Loss 2000-2014 (*r*)
Nigahau	10.44	194	1858.2	-0.67	3.64	0.76	-0.015	-0.008	-0.012
Hesesai	23.82	171	717.9	-	-1.10	-	-0.034	-0.002	-0.018
Dadahai	27.00	37	137.1	-1.18	3.53	0.27	-0.018	-0.001	-0.009
Grass	163.64	448	273.8	0.19	8.56	3.45	-0.013	-0.024	-0.018
Nimowa	353.59	481	136.0	5.97	7.14	8.97	-0.004	-0.009	-0.007
Sabara	405.08	640	158.0	2.44	0.86	1.86	n/a**	n/a**	n/a**
Hemenahei*	1014.71	-	-	-	-	-	-0.001	-0.003	-0.002
Panawina	2930.93	578	19.7	1.49	6.00	3.85	-0.001	-0.002	-0.002
Junet	7698.30	368	4.8	10.34	-3.46	2.48	-0.002	-0.005	-0.003
Sudest	80860.23	3780	4.7	2.37	1.41	2.26	-0.001	-0.001	-0.001
Ward 4	8964.80	559	6.2	1.31	2.55	2.03	-0.002	-0.003	-0.002
Ward 5	8238.48	762	9.2	2.59	1.89	2.70	-0.003	-0.003	-0.003
Ward 6	24491.70	811	3.3	5.49	0.69	3.97	-0.001	-0.001	-0.001
Ward 7	17176.10	763	4.4	3.89	1.43	3.39	-0.001	-0.001	-0.001
Ward 8	12684.40	372	2.9	0.30	0.57	0.42	-0.001	-0.001	-0.001
Ward 9	4319.20	331	7.7	0.58	2.36	1.35	-0.001	-0.002	-0.002
Ward 16	5097.22	182	3.6	0.46	0.14	0.34	-0.001	-0.001	-0.001

*This island supports a village (Niyelahoi) and small hamlets (at time of study) on the north and north-western sides but these appear to have been incorporated into surrounding island village census data. Source: Local government census data books.
**Sabara Island is not used for gardening and forest loss rates were not calculated due to zero loss. Forest loss rates (*r*) were calculated following Puyravaud (2003).

Table S2. Full results using the Dredge function of model estimates of predictor variables for forest loss (MuMIn package; Bartoń, 2016).

Intercept	Autocov	E	FC	DV	E:FC	E:DV	FC:DV	E:FC:DV	df	logLik	AIC	ΔAIC	weight
-1.399	1.127	0.197	-0.317	0.004	-0.431	-0.081	0.102	0.229	10	-8655.942	17331.884	0	1
-1.479	1.169	0.145	-0.177	0.059	-0.279	-0.118	-	-	8	-8710.219	17436.438	104.554	1.98E-23
-1.477	1.169	0.144	-0.181	0.060	-0.277	-0.118	-0.007	-	9	-8710.184	17438.368	106.484	7.54E-24
-1.481	1.141	0.097	-0.127	0.047	-0.234	-	-	-	7	-8728.113	17470.227	138.343	9.10E-31
-1.482	1.117	0.106	-0.130	-	-0.239	-	-	-	6	-8729.556	17471.112	139.228	5.85E-31
-1.479	1.140	0.096	-0.132	0.047	-0.232	-	-0.009	-	8	-8728.061	17472.122	140.238	3.53E-31
-1.594	1.210	0.109	-0.056	0.099	-	-0.060	-0.073	-	8	-8757.232	17530.465	198.581	7.56E-44
-1.627	1.217	0.112	-	0.094	-	-0.059	-	-	6	-8761.586	17535.172	203.288	7.19E-45
-1.630	1.216	0.113	-0.006	0.095	-	-0.060	-	-	7	-8761.530	17537.061	205.177	2.80E-45
-1.624	1.196	0.083	-0.045	0.091	-	-	-0.073	-	7	-8762.720	17539.440	207.556	8.51E-46
-1.663	1.202	0.088	-	0.088	-	-	-	-	5	-8766.934	17543.867	211.983	9.30E-47
-1.661	1.203	0.088	0.004	0.087	-	-	-	-	6	-8766.905	17545.809	213.926	3.52E-47
-1.540	1.141	0.107	-	-	-	-	-	-	4	-8771.777	17551.554	219.670	1.99E-48
-1.631	1.187	-	-0.041	0.116	-	-	-0.079	-	6	-8770.074	17552.147	220.263	1.48E-48
-1.540	1.144	0.106	0.010	-	-	-	-	-	5	-8771.627	17553.254	221.371	8.51E-49
-1.678	1.189	-	-	0.114	-	-	-	-	4	-8775.315	17558.629	226.746	5.79E-50
-1.671	1.193	-	0.013	0.113	-	-	-	-	5	-8775.070	17560.140	228.256	2.72E-50
-1.683	1.120	-	-	-	-	-	-	-	3	-8784.027	17574.053	242.169	2.59E-53
-1.673	1.126	-	0.017	-	-	-	-	-	4	-8783.589	17575.178	243.295	1.48E-53
-1.121	-	0.174	-0.719	-0.748	-0.726	-0.003	0.151	0.390	9	-9642.001	19302.003	1970.119	0
-1.248	-	0.067	-0.524	-0.700	-0.525	-0.085	-0.042	-	8	-9775.382	19566.763	2234.879	0
-1.264	-	0.071	-0.502	-0.703	-0.538	-0.088	-	-	7	-9776.626	19567.252	2235.368	0
-1.309	-	0.050	-0.491	-0.693	-0.484	-	-0.050	-	7	-9782.392	19578.784	2246.900	0
-1.330	-	0.053	-0.464	-0.696	-0.497	-	-	-	6	-9784.081	19580.162	2248.278	0
-1.476	-	-0.003	-0.266	-0.685	-	0.057	-0.154	-	7	-9929.329	19872.657	2540.774	0
-1.442	-	-	-0.271	-0.685	-	-	-0.151	-	5	-9933.376	19876.752	2544.868	0
-1.442	-	0.004	-0.272	-0.686	-	-	-0.151	-	6	-9933.355	19878.710	2546.827	0
-1.556	-	-0.003	-0.160	-0.689	-	0.052	-	-	6	-9949.590	19911.179	2579.295	0
-1.525	-	-	-0.167	-0.689	-	-	-	-	4	-9952.604	19913.207	2581.323	0
-1.525	-	0.001	-0.167	-0.690	-	-	-	-	5	-9952.601	19915.203	2583.319	0
-1.456	-	-0.036	-	-0.727	-	0.087	-	-	5	-9992.377	19994.753	2662.870	0
-1.396	-	-0.034	-	-0.732	-	-	-	-	4	-10000.652	20009.304	2677.420	0
-1.386	-	-	-	-0.745	-	-	-	-	3	-10001.922	20009.844	2677.960	0
-0.958	-	-0.159	-0.573	-	-0.506	-	-	-	5	-10263.161	20536.321	3204.438	0
-1.147	-	-0.189	-0.268	-	-	-	-	-	4	-10433.048	20874.096	3542.212	0
-1.068	-	-	-0.303	-	-	-	-	-	3	-10479.632	20965.264	3633.380	0
-0.879	-	-0.261	-	-	-	-	-	-	3	-10561.193	21128.386	3796.503	0
-0.710	-	-	-	-	-	-	-	-	2	-10650.629	21305.259	3973.375	0

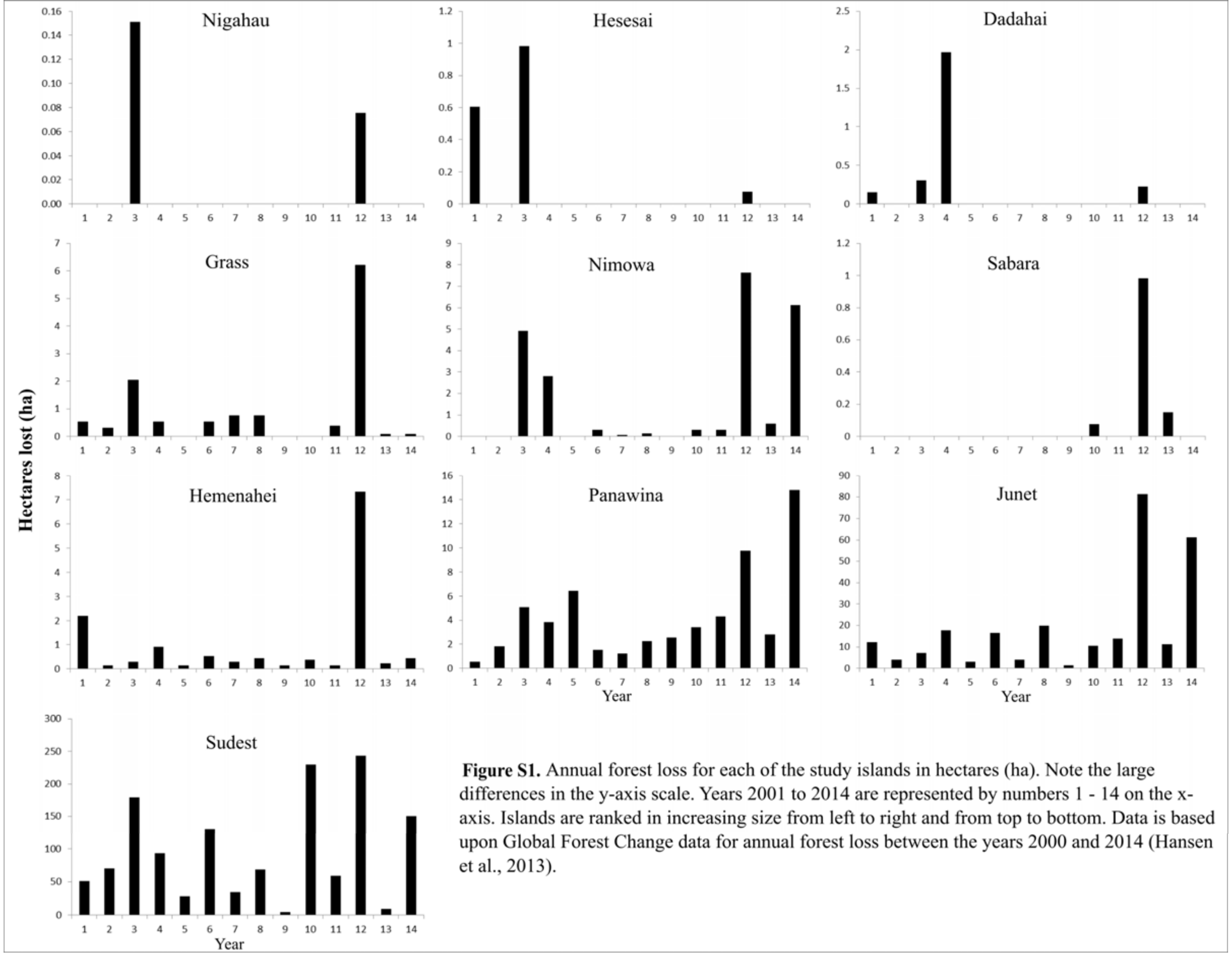

Figure S1. Annual forest loss for each of the study islands in hectares (ha). Note the large differences in the y-axis scale. Years 2001 to 2014 are represented by numbers 1 - 14 on the x-axis. Islands are ranked in increasing size from left to right and from top to bottom. Data is based upon Global Forest Change data for annual forest loss between the years 2000 and 2014 (Hansen et al., 2013).

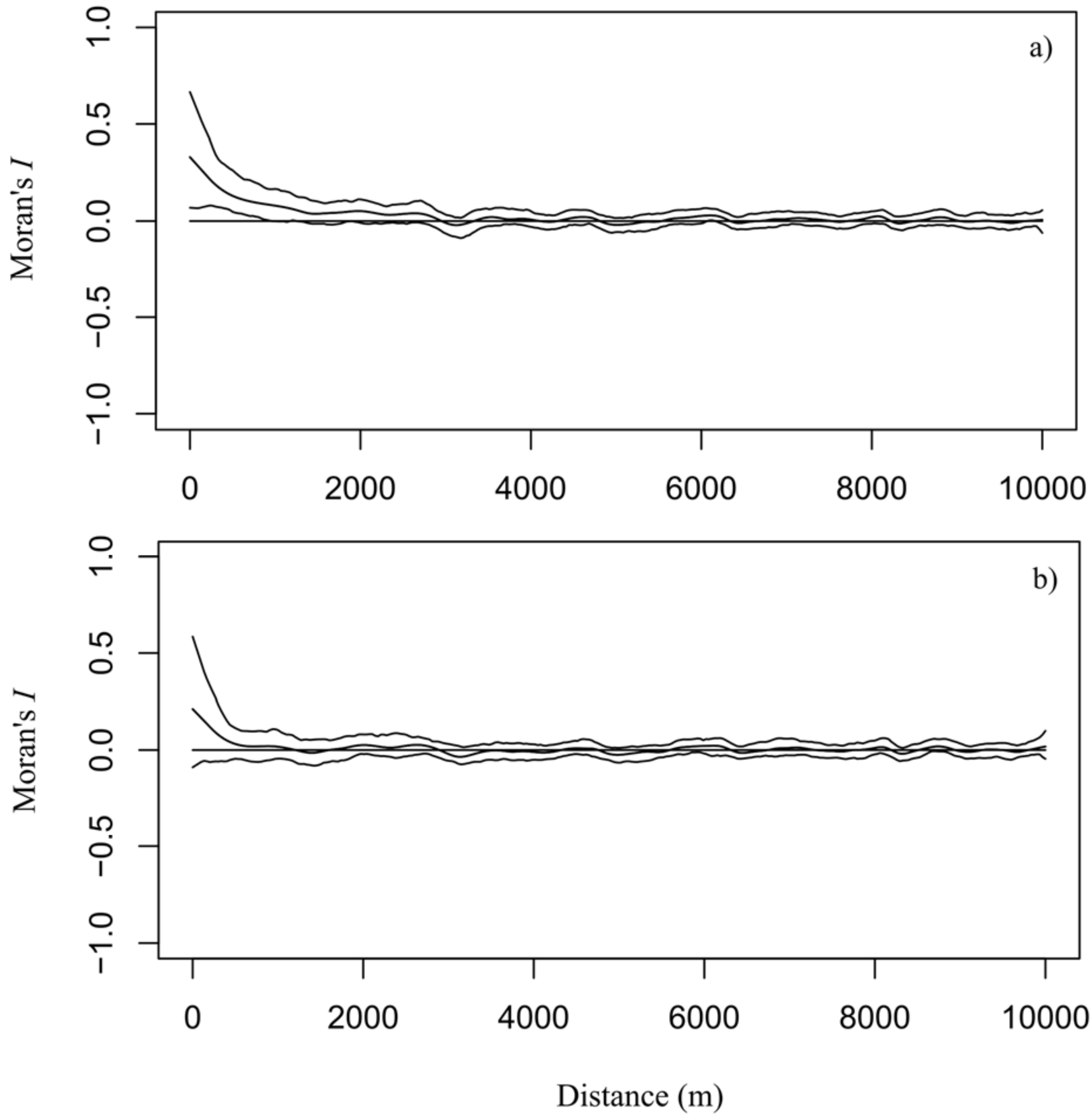

Figure S2. Correlograms of forest loss model residuals a) before the introduction of an autocovariate and b) the final model with an autocovariate introduced. Both show the Moran's *I* (observed) bounded by the upper and lower 95% confidence intervals. The slight model improvement is most apparent at very near distances where spatial autocorrelation occurred in a).