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Comparing field-based monitoring and remote-sensing, using deforestation from logging at Important Bird Areas as a case study

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ABSTRACT

Monitoring sites of biodiversity conservation importance is essential for their conservation. It allows threats to be identified and quantified, priorities to be set, responses to be developed, and facilitates adaptive management. Field-based monitoring protocols need to be simple enough to be widely applied in countries with limited capacity while being sufficiently robust to provide widely reliable data. A simple, globally standardised monitoring protocol is now being implemented at thousands of sites of global avian conservation significance (Important Bird Areas, IBAs) worldwide, but the consistency of the approach across sites, countries and regions remains untested. We tested the match between estimates of the threat to IBAs from logging derived from such monitoring, with standardised deforestation rates derived from remote sensing data for 2000–2005 to determine if the two were consistently related. We found a significant positive correlation between the impact of the threat from logging and the proportion of forest lost (although the gross forest loss did not differ systematically with the two components of the threat impact: scope and severity). The results give us some confidence that the simple field-based protocol being implemented by a diversity of surveyors with varied technical capacity can generate meaningful and consistent monitoring data across the globe.

1. Introduction

Monitoring is essential to identify threats to biodiversity, to develop appropriate responses to threats, and to assess the effectiveness—and facilitate adaptive management—of ensuing actions (Jones et al., 2013). For sites of conservation significance, such as protected areas, monitoring should ideally evaluate the state (condition) of conservation targets (populations, species, habitats, etc.), the pressures (threats) affecting them, and the conservation actions (responses) in place. While remote sensing can be useful for monitoring trends in land cover extent, at least for forest (e.g. Achard et al., 2002; Hansen et al., 2010), it is less useful for evaluating trends in habitat condition and population abundance. Similarly, while it may be useful for monitoring some threats (e.g. deforestation for oil palm plantations (e.g. Buchanan et al., 2008), it cannot directly measure hunting pressures. Hence, remote sensing often requires complementary field-based monitoring to provide adequate data for informed and comprehensive decision-making and adaptive management. However, capacity to undertake such monitoring is limited, particularly in the poorest countries which often have the richest biodiversity and most intense threats.

To address this, BirdLife International has developed a simple, globally standardised protocol for monitoring Important Bird Areas, or IBAs (BirdLife International, 2006). IBAs are sites of global significance for bird conservation, identified nationally through multi-stakeholder processes using globally standardised criteria for populations of threatened, restricted-range, biome-restricted or congregatory species (BirdLife International, 2012). Over 12,000 have been identified worldwide (BirdLife International, 2013), and IBA monitoring is underway at over 25% of these. The IBA monitoring protocol focuses on evaluating the state (condition of), pressures (threats to), and response (management, mitigation, etc.) at IBAs, each of which is scored on a simple 4-point scale (e.g. Mwangi et al., 2010). It is typically implemented by staff of BirdLife partner organisations, or members of local community conservation groups developed and supported by these partners, other volunteers, or protected-area staff. These people are given basic training, but many have relatively low technical capacity.

The consistency of the approach across sites and countries has never been tested, partially due to the absence of an appropriate, globally referenced dataset that could relate to aspects of the field assessments. Hansen et al. (2010) produced a coarse resolution map of gross forest loss between 2000 and 2005 from remote sensing data at a spatial resolution of 18.5 km by 18.5 km cells.

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To assess comparability between the two approaches, we matched threat impact scores derived from field monitoring for a sample of IBAs with deforestation estimates based upon Hansen et al. (2010) for these sites.

2. Methods

2.1. Field based monitoring

Data derived from field-based monitoring assessments at IBAs were extracted from BirdLife's World Bird Database (WBDB). Following a standardised protocol (BirdLife International, 2006), scores for state (condition), pressure (threats) and responses (action) are determined for each site. To evaluate pressure, the principal threats affecting those species for which the site was identified as an IBA (i.e. the 'trigger species') are classified using the IUCN-CMP[1] Threats Classification Scheme (Salafsky et al., 2008; http://www.iucnredlist.org/technical-documents/classification-schemes/threats-classification-scheme). Each threat is then scored for its impact, derived from separate assessments of timing, scope, and severity of each threat type. Scope is defined as the proportion of the population—or the proportion of the key habitat for each trigger species—affected, with scores assigned as 0 (<10%), 1 (10–49%), 2 (50–90%) or 3 (>90%). Severity is defined as the maximum rate of population decline of any trigger species caused by the threat within its scope, with scores assigned as 0 (<1% over 10 years or three generations of the trigger species, whichever is longer), 1 (1–10% over the same period), 2 (11–30% over 10 years) and 3 (>30% over 10 years). Timing is scored on the basis of whether each threat is past and no longer limiting (score 0), likely in the long term i.e. >4 years (1), likely in the short term or <4 years (2) or happening now (3). The overall threat impact is then calculated from the sum of these scores. This resultant 0–9 scale can be converted back to a four-point scale of 0–3 (with 3 indicating the highest level of threat). The threat or threats with the highest score represent the overall threat impact score for the site, following a 'weakest link' approach (for further details see BirdLife International, 2006; Mwangi et al., 2010).

These scores cover a wide range of threats (Salafsky et al., 2008), but we were interested solely in the IBAs that were identified for forest species and where "logging" was identified as a threat "happening now". We identified IBAs identified for species estimated to have "high" or "medium" dependency on forests (Buchanan et.al. 2011). Species with high forest-dependence are forest specialists that are characteristic of the interior of undisturbed forest, rarely occupy non-forest habitats, and almost invariably breed within forest; while they may persist in secondary forest and forest patches if their particular ecological requirements are met, they are usually less common in such situations. Species with medium forest-dependence are forest generalists that breed in undisturbed forest but are also regularly found in forest strips, edges and gaps and secondary forest, where they may be commoner than in the interior of intact forest. There were 96 IBAs that had been monitored between 2000 and 2005 and that had a threat of logging to forest-dependent species documented as "happening now". The mean size of these IBAs was 297.4 km^2 (median = 381.3 km^2) and 45 were smaller than 342 km^2 (the area of a cell used by Hansen et al. (2010)).

2.2. Remote sensing data

While a range of land-cover maps from multiple time periods are available, only Hansen et al. (2010) has been produced in such a way that enables appropriate comparisons of land-cover change over time in a way that is comparable worldwide. These authors estimated forest cover and gross forest loss between 2000 and 2005 in 18.5 km by 18.5 km cells across most of the globe (many oceanic islands excepted) using MODIS satellite image data at 500 m resolution, calibrated by analysis of higher resolution Landsat data. Digital boundaries were obtained for 10,253 IBAs (BirdLife International, 2011), representing c. 95% of globally significant IBAs identified by 2011. Boundaries for many IBAs in USA, and for small numbers in other countries, were unavailable, while IBAs had not yet been identified in New Guinea and several other island Pacific states at the time of the analysis. The forest loss and IBA maps were projected using Berhmann's equal-area projection. IBAs smaller than 1 km^2 were scaled up to 1 km^2, based on the Berhmann projection. While some 7895 of these IBAs overlapped with the deforestation map, only 4491 were designated for forest species. The median size of these selected IBAs was 223 km^2 (range 1–73,511 km^2) while 2669 IBAs (59%) had an area less than 342 km^2 (the cell size used by Hansen et al. (2010). Mean and maximum proportional forest loss within each IBA were determined by overlaying. Digital IBA boundaries were overlaid onto the Hansen et al. (2010) deforestation map in ArcMap 10, and the percentage forest loss was calculated for each 18.5 km square cell used by Hansen et al. (loc. cit.) that partially or wholly overlapped each IBA. We then determined the mean percentage forest loss for all cells overlapping each IBA, and the maximum percentage loss for any cell overlapping the IBA. Due to the mismatch in size of IBAs and spatial unit of forest loss, these estimates are best described as loss in the location of IBAs. Additionally, mean forest loss in the cells immediately surrounding, but not overlapping each IBA (i.e. in the neighbouring locations) was calculated.

2.3. Analysis

To compare deforestation estimates from Hansen et al. (2010) with the impact of logging as assessed on the ground, ranks were calculated for each IBA based on (a) threat impact score (severity score plus scope score) from field monitoring, and (b) mean and (c) maximum forest loss (from remote sensing). We compared these ranks using a Spearman rank correlation. ANOVAs using generalised linear models (in Proc Genmod in SAS 9.2) were used to test for systematic differences between IBAs in threat scores for scope, severity and impact. Mean and maximum forest losses in IBA locations was the dependant variable, and logging scope, severity and impact scores were entered as fixed effects. Log transformation was used to normalise the forest loss data for these GLMs. The significance of relationships was assessed by comparing the explained deviance with the χ^2 distribution using the appropriate degrees of freedom (e.g. Crawley, 1993). Residuals were examined to identify IBAs where loss fell outside the 95% CI around the mean for focal scores. Forest loss in the location of IBAs was compared to loss in cells in the neighbouring locations using a sign test. All means presented ± SE.

3. Results

There was a significant positive correlation between the threat impact score for IBAs and both mean and maximum forest loss (rho = 0.227, P = 0.026 and rho = 0.273, P = 0.007 respectively; Fig. 1) in the location of IBAs for the 96 IBAs for which these data were available. The significant correlations across all IBAs indicates that there was a broad agreement between remote sensing and field monitoring estimates of the intensity of pressures from forest loss. However, neither the mean nor maximum rate of loss differed systematically with threat impact score in an ANOVA ($\chi^2_6 = 6.07$,

[1] Conservation Measures Partnership.