# Management resourcing and government transparency are key drivers of biodiversity outcomes in Southeast Asian protected areas


## Abstract

Protected areas aim to conserve nature by providing safe havens for biodiversity. However, protection from habitat loss, poaching and other threats, is not guaranteed without adequate investment in their management. Here, we examine the relationship between management effectiveness using the Management Effectiveness Tracking Tool (METT) and trends of 79 populations of mammals and birds in 12 Southeast Asian protected areas from Cambodia, Indonesia, Thailand and Vietnam. Despite the negative influence of corruption on species population change, we find evidence that adequate financial and human resourcing are important determinants in achieving good biodiversity outcomes. Management resourcing, national government transparency and body size collectively explain 29% of the variation in animal population trends in our model. Our paper contributes to a growing evidence base linking management resourcing shortfalls to declining biodiversity populations in protected areas. Our key findings are relevant to international funding agencies, governments and NGOs, to aid decision making around the allocation of conservation resources in Southeast Asia.



**Keywords**: protected areas, biodiversity, animal population trends, management effectiveness tracking tool (METT), government transparency, Living Planet Database (LPD), Southeast Asia


## 1. Introduction

Protected areas are an essential tool for conserving nature, ecosystem services and cultural values (UNEP-WCMC 2018). Despite a tripling in size of land set aside to conserve nature over the past 40 years, biodiversity is continuing to decline (Watson et al. 2014). Ecological communities worldwide have lost 20% of originally-present, terrestrial species (IPBES 2019), and population sizes of vertebrates have declined by 68% on average between 1970 and 2016 when controlling for taxonomic biases (WWF 2020). As we approached the end of the United Nations Strategic Plan for Biodiversity 2011–2020 (CBD 2011), reflections on the adequacy of global conservation targets and progress towards achieving them, highlight that bolder area coverage targets are needed for the post-2020 decade (Allan et al. 2019; Jones et al. 2019; Woodley et al. 2019). However, creating any number of new protected areas will have minimal impact on biodiversity conservation without adequate resources dedicated to the ongoing management of threats (Coad et al. 2019). Therefore, of equal importance, is reflection on the effectiveness of protected areas, captured by the part of the Convention on Biological Diversity Aichi Target 11 that calls for effective management.

Since the first global review of Protected Area Management Effectiveness (PAME) in 2010 (Leverington et al. 2010), scientists have raised attention to the need for unified, quantitative metrics of protected area effectiveness (Coad et al. 2019; Geldmann et al. 2018; Geldmann et al. 2019). The International Union for the Conservation of Nature (IUCN) Green List Standard is widely recognized as the new global standard for assessing whether protected areas are achieving conservation outcomes through effective management and equitable governance (IUCN and WCPA 2017). However, because it is new, it has not yet been widely applied in protected area evaluations. The Management Effectiveness Tracking Tool (METT; Stolton et al. 2007) is the largest global collation, and the official repository, of information on management effectiveness data for all signatories to the Convention on Biological Diversity (CBD) and a requirement of all Global Environment Facility funded-projects (Coad et al. 2015; Coad et al. 2019). Park managers are required to use the best available evidence and their expert judgement to complete the assessments based on a comparable and standardized framework for all sites. Though not without their biases and limitations (i.e. subjectivity), METT responses have been found to be good indicators of on-ground park realities in Australia (Cook et al. 2014), and have been used to build evidence that global-scale under-resourcing of protected areas is linked to biodiversity declines, in both terrestrial and marine realms (Geldmann et al. 2018; Gill et al. 2017). Protected areas have reduced rates of biodiversity loss compared to unprotected sites (Geldmann et al. 2013), yet with significant variance between sites (Barnes et al. 2016; Beaudrot et al. 2016). Exploration of the managerial and socioeconomic conditions that are most important for effectively managing biodiversity inside protected areas is critical to understanding why protected areas are (or are not) delivering on their intended outcomes of protecting biodiversity (Barnes et al. 2016, Geldmann et al. 2018).

Further, location biases exist in data quantity for protected areas, with more comprehensive data from Europe and North America. Therefore, the extent that these global findings relate to regional or local dynamics is unknown. The global biodiversity hotspot of Southeast Asia (Myers *et al.* 2000) has little representation in global terrestrial studies, despite the region experiencing one of the highest intensities of human pressure (Venter et al. 2016), and rapid biodiversity declines and extinctions of large-bodied fauna even in intact forests (Benítez-López et al. 2019). Lack of clear evidence on the effectiveness of conservation interventions is a research gap reported by Southeast Asian conservation practitioners following several failed interventions aiming to prevent the local extinction of critically endangered species (Coleman et al. 2019).

Here, we explore how management resourcing affects biodiversity trends in Southeast Asian protected areas. We apply a model to test the strength of the relationship between animal population trends from the Living Planet Index Database (LPD 2018) and a select group of management factors and contextual factors that are known, or predicted, to influence biodiversity conservation in terrestrial protected areas. Measuring the impact of protection on biodiversity requires comparison with a similar, but unprotected site (the counterfactual). Lack of long-term monitoring of biodiversity outside protected areas prohibits large-scale studies using counterfactual design approaches. However, correlational studies, like this, are suitable for identifying broad patterns between managerial and socioeconomic conditions and biodiversity population trends.

## 2. Methods

### *2.1. Protected area management effectiveness*

We collated surveys of management effectiveness of terrestrial protected areas from the Management Effectiveness Tracking Tool (METT). We developed an approach that aligns the METT criteria with the four IUCN GreenList components based on congruence between objectives measured by each indicator (Table S1). We used METT assessments conducted between 2000 and 2014 as measures of protected area management effectiveness. Each METT assessment consists of 30 questions that are scored from 0 (inadequate) to 3 (adequate). We used the following approach to select, exclude and re-align METT survey responses to our predictors of interest. For protected areas with multiple assessments over time, we considered the oldest assessment to be temporally appropriate, as management interventions should precede any resulting biodiversity outcomes. We removed METT questions that were not directly linked to biodiversity in the short-term. We also excluded the conservation outcomes survey responses and replaced them with ‘animal population trends’ (see below). The effective management component had more questions than any other category, therefore we split it into two sub-categories: management resourcing and management processes based on the IUCN GreenList components (see Appendix S1 for details). Management resourcing included questions on the implementation of management objectives (Q4), management plan (Q7), work plan (Q8), staff numbers (Q12), budget (Q15) and equipment (Q18). Management processes included questions on information availability to manage the area (Q9) and its design (Q5). We tested for collinearity between responses by performing Spearman rank correlations. This led to the exclusion of seven covariates (see S1). We were left with the following four dimensions of management: (1) good governance, (2) sound design and planning, (3) management resourcing, and (4) management processes. Finally, we calculated an average score for all METT questions within each of these four groups.

### *2.2. Animal population trends*

We obtained animal population time-series data from the Living Planet Index Database, which collates data from published manuscripts, online databases and grey literature (LPD 2018), and from correspondence with local experts. We included population records for terrestrial species only as population trends in marine species are expected to be more directly influenced by the management of marine, rather than terrestrial, protected areas (which had been excluded from the outset). In the Living Planet Index Database, a population is a set of individuals of a species that is monitored in a consistent way over time in the same location. Using this definition, any bird (or other vertebrate) is deemed to be in a protected area if the monitoring was done entirely within the park boundaries, irrespective of how much time it spends there normally or to what extent this protected area forms part of its range. Populations included in the index must meet certain time-series criteria to improve certainty that these populations, especially the more mobile ones (e.g. migratory birds), are more than occasional visitors. Following Geldmann et al. (2018), only populations that had a minimum of three observations were considered, but we adopted a more restrictive inclusion criteria that observations had to extend over at least a five-year period in order to reflect the management effectiveness. Following Barnes et al. (2016), we excluded all records of “zeros” that were not indicative of a population going extinct. For all records that passed these selection criteria, trends in animal populations were calculated as the annual rate of change over time (ie the slope) by fitting a linear regression model to the scaled population values, following Barnes et al. (2016) and Geldmann et al. (2018; Appendix S1).

### *2.3. Final dataset*

The final dataset was restricted by the availability of matching METT data and biodiversity population data from terrestrial protected areas (UNEP-WCMC 2018). We augmented these datasets with data directly supplied by local experts (resulting in 20 extra populations and 1 extra METT from 7 sites). Our dataset, comprised of 79 populations with population values measured between 1965-2018, encompassing 55 species (Table S2), from 12 terrestrial protected areas and four countries, reflects the most temporally appropriate sources for this analysis, which may not reflect current conditions in the protected area.

### *2.4. Statistical modelling approach*

We built a predictive linear model that tests the direction and strength of the relationship between animal population trends, management factors and contextual factors (Figure 1). We considered all key factors that are known or predicted to influence biodiversity in terrestrial protected areas. They include geographic biases (elevation, accessibility), size and age of the protected area, forest cover loss, perceived national government transparency and animal’s body mass (Barnes et al. 2016, Geldmann et al. 2018). Village-level wealth and population density metrics were not available at an appropriate spatial-scale. In our model, the annual rate of change (ie the slope) for each of the 79 animal populations was our dependent variable and the four management factors, as well as: (1) time protected, (2) protected area size (3) accessibility (to the nearest city), (4) elevation, (5) body size, (6) national government transparency, and (7) forest cover loss, were used as independent variables (Table 1; S1).  We chose to run the model on species populations so that we could detect any variation in how different populations respond to management (i.e. larger species may be slower to recover from management efforts or face more severe threats). The best-fit model was determined based on Akaike Information Criterion (AIC) and $R^2$ total variance explained of all possible configurations of predictor variables (Appendix S1), using the MuMIn package (Barton & Barton, 2015). Finally, we conducted post-hoc correlation analysis to identify the specific management variables that best explained the variation in animal population trends. All spatial analysis was performed in ArcGIS v10.5 (ESRI 2016) using the Asia South Albers Equal Area Conic projection. Statistical modeling was performed in R v3.4.3 (R Development Core Team 2017).

## 3. Results

### *3.1. Data coverage*

Across the region of Southeast Asia, there are 1,376 designated protected areas officially included in the World Database of Protected Areas (IUCN 2018), covering 549,990km$^2$ (14% of the region). The total overlap between 118 population time-series from 23 protected areas and the METT assessments comprised data of 79 populations from 12 terrestrial protected areas (Figure 2), after applying the exclusion criteria. Geographically, our dataset had protected areas from Cambodia (n = 5), Indonesia (n = 3), Vietnam (n = 3), and Thailand (n = 1). Taxonomically, our biodiversity time-series was mostly for mammals (n = 53, 67%), rather than birds (n = 26, 33%), over a monitoring period from 1965 to 2018. Amphibians, invertebrates and reptiles did not have long-term published datasets. Our sample was biased towards older and larger protected areas: the median age from our sample was 33 years compared to the regional median of 26 years; and the median size from our sample was 2,377km$^2$ compared to the regional median of 59km$^2$. However, our sample was not biased towards protected areas with more positive animal population trends (the mean rate of change from our sample: 4.15%, versus all biodiversity from protected areas in the LPD: 4.93%).

### *3.2. Model outcomes*

Our best performing model, based on AIC, showed that overall population trends (n = 79) in Southeast Asian protected areas are best explained by management resourcing, government transparency, and body size, with no interaction effects (F-statistic: 7.478, P: 1.033e-05; Adjusted R-squared: 0.293; Figure 3). Management resourcing and government transparency had a significant positive relationship with biodiversity populations. Body size had a significant negative relationship with biodiversity populations. Post-hoc exploration of the dimensions of management resourcing identified that adequate financial resourcing ($\rho$ = 0.51), human resourcing ($\rho$ = 0.42) and equipment ($\rho$ = 0.26) had the strongest positive relationships with biodiversity outcomes (Spearman rank correlation). Staff training and budget management were highly correlated with these variables. Our best performing model also included good governance of protected areas and forest cover loss, but these two variables did not have significant relationships with animal population trends.

## 4. Discussion

We found that adequate management resourcing (financial, human and technological capacity) and government transparency are associated with more positive rates of change for animal populations inside Southeast Asian protected areas, and body size is associated with more negative rates of change. Management resourcing, national government transparency and body size collectively explained one-third of the model variation. By combining time-series biodiversity data with protected area management effectiveness surveys and socio-economic indicators, our analysis provides evidence that positive animal population trends are associated with higher levels of management resourcing, and that the relationship is stronger in less corrupt countries. This is consistent with the results from a global study (Geldmann et al. 2018), where representation from Southeast Asia was low (48 populations from 4 protected areas). Our paper provides preliminary evidence using a richer dataset (79 populations from 12 protected areas) that this pattern also holds true in the Southeast Asian countries we sampled. We can hypothesise that with more representation from countries with lower levels of corruption, this strength of this pattern would increase and management resourcing will have a more pronounced influence on animal population trends. Data from Southeast Asia is limited by resourcing constraints, data quality issues and limited access to the available data.

Our finding that smaller animals showed more positive population trends, differs to a global study that reports the opposite relationship (Barnes et al. 2016). We expect the causal mechanism underlying this pattern is the high prevalence of poaching in Southeast Asia, which has historically targeted large animals (e.g. rhinoceros, elephant, tiger) of high economic value as trophies and medicine, and is a major driver of regional biodiversity declines (Harrison et al. 2016; Steinmetz et al. 2010). Contrary, global studies have been dominated by African protected areas where the preservation of larger iconic mammals can contribute significantly to the national economy through tourism (Naidoo et al. 2016). Our results are corroborated by evidence from the ground. First, management staff from Cat Tien National Park in Vietnam flagged in a 2003 METT survey that inadequate capacity and resources were negatively affecting their ability to meet the park's management objectives, including the protection of a flagship species, the Javan rhinoceros (*Rhinoceros sondaicus*). In 2010, the last Javan rhinoceros in Cat Tien National Park was poached marking its local extinction from Vietnam. A published review found that the failure to protect this species from extinction was tied to insufficient patrol staff for the area, inadequate capacity and monitoring resources, exacerbated by a poorly regulated market in Vietnam for rhino horn (Brook et al. 2014). Staff from Bukit Barisan Selatan National Park in Indonesia reported in a 2003 METT survey they had insufficient human and financial resources to patrol the 3,168km$^2$ former safe haven for the Sumatran rhinoceros (*Dicerorhinus sumatrensis*), which was potentially compounded by corruption. The species has since suffered rapid declines to the point of its disappearance and probable functional extinction (Hance 2019). In contrast, financial support for patrol staff, and community support from village volunteers was linked to the recovery of several populations of Gaur (*Bos gaurus*), Wild boar (*Sus scrofa*), and Red muntjac (*Muntiacus muntjak*) that were severely hunted in Thung Yai Wildlife Reserve in Thailand until 1995 (Steinmetz et al. 2010). Similarly, monitoring data from Huai Kha Khaeng Wildlife Sanctuary in Thailand also shows that tiger survival rates and recruitment increased following efforts of intensified patrolling from 2006 to 2012, though population recoveries were slow. The latter two examples highlight the potential for small populations to recover if management efforts are scaled-up in response to increases in threat intensity and pressure (Geldmann et al. 2019).

Our model did not detect a link between forest cover loss and animal population trends in protected areas. However, our result does not infer that there is no link between biodiversity and deforestation, as there is conclusive evidence that habitat loss drives biodiversity declines at a global scale (Brooks et al. 2002). Instead, we highlight some ecological, social, and technical factors that limit the ability of remote-sensing derived forest cover maps to represent animal population trends in tropical forest ecosystems, consistently across space and time. Firstly, species have variable levels of resilience to habitat change, and not all species in tropical forests are forest-dependent (Ewers and Didham 2006). Even for forest-dependent species, abundance does not have a linear relationship with forest cover (Green et al. 2020). There is also a lag-effect before biodiversity declines are fully realized after environmental perturbations, known as extinction debt (Kuussaari et al. 2009). Secondly, even in some intact tropical forests across Southeast Asia, large mammals are absent due to poaching (Benítez-López et al. 2019). Finally, some level of classification error arises when using remote-sensing techniques to produce tree cover maps, as land-use changes from natural forest to plantation forests cannot always be detected (Sexton et al. 2016). Our sample did not contain any time-series data of reptiles or amphibians, which is a representation of real biases that exist in sampling effort, which tend to favour mammals and birds. Similar biases are likely to exist in geographic terms, which may favour political or tourism priorities. If we had a larger sample size, the data might allow us to explore more national and local predictors, such as wealth and population density. The ability to produce conclusive inference on the patterns between protected area management and conservation outcomes is severely constrained by inherent issues with both management effectiveness and biodiversity time-series data, that has been extensively discussed in the literature (Geldmann et al., 2018; 2019). Our study brings to light new evidence that addresses the ongoing debate on how to allocate resources to better protect nature (Adams et al. 2019). For over two decades, evidence linking under-funding to species declines and extirpations has grown; highlighting that conservation spending needs to be scaled-up. From within the pool of resources spent on nature conservation globally, biodiversity hotspots , such as Indo-Burma, Sundaland, the Philippines, and Wallacea in Southeast Asia, require more conservation investment as they have a large share of globally threatened biodiversity (Myers et al. 2000; Rodrigues et al. 2004). Despite warnings that developing country hotspots need prioritized investment (Balmford et al. 2003), only 6% of total global conservation expenditure ($21.5 billion USD allocated globally) went to low and middle income countries for the 2001-2008 period (Waldron et al. 2013). Socioeconomic context can undermine conservation efforts in developing countries with high poverty rates, causing concern that conservation spending may fail to trigger any real, lasting impact. However, despite the negative influence of corruption on conservation investment priorities, it has less influence than purchasing power parity when investing in low income, developing countries and less importance than investing in countries with more single site threatened species (Garnett et al. 2011).

Fostering investor confidence in the likelihood of conservation outcomes is important to mobilising more financial support for developing countries. Confidence can be strengthened by building a geographically diverse evidence base that links biodiversity outcomes to management effort. As a more compelling evidence base builds, it may ultimately persuade decisions-makers to implement bolder steps to achieve international and regional commitments to stop species extinctions and declines, by scaling-up investment for nature conservation globally, but especially in developing countries. Correlational studies, like this, are crucial to collating evidence on the links between biodiversity, protected area management resourcing and socioeconomic factors. By focusing specifically on a developing region that is under-represented in global biodiversity and protected area effectiveness datasets, yet with a large share of globally threatened biodiversity, we attempted to address this evidence gap.

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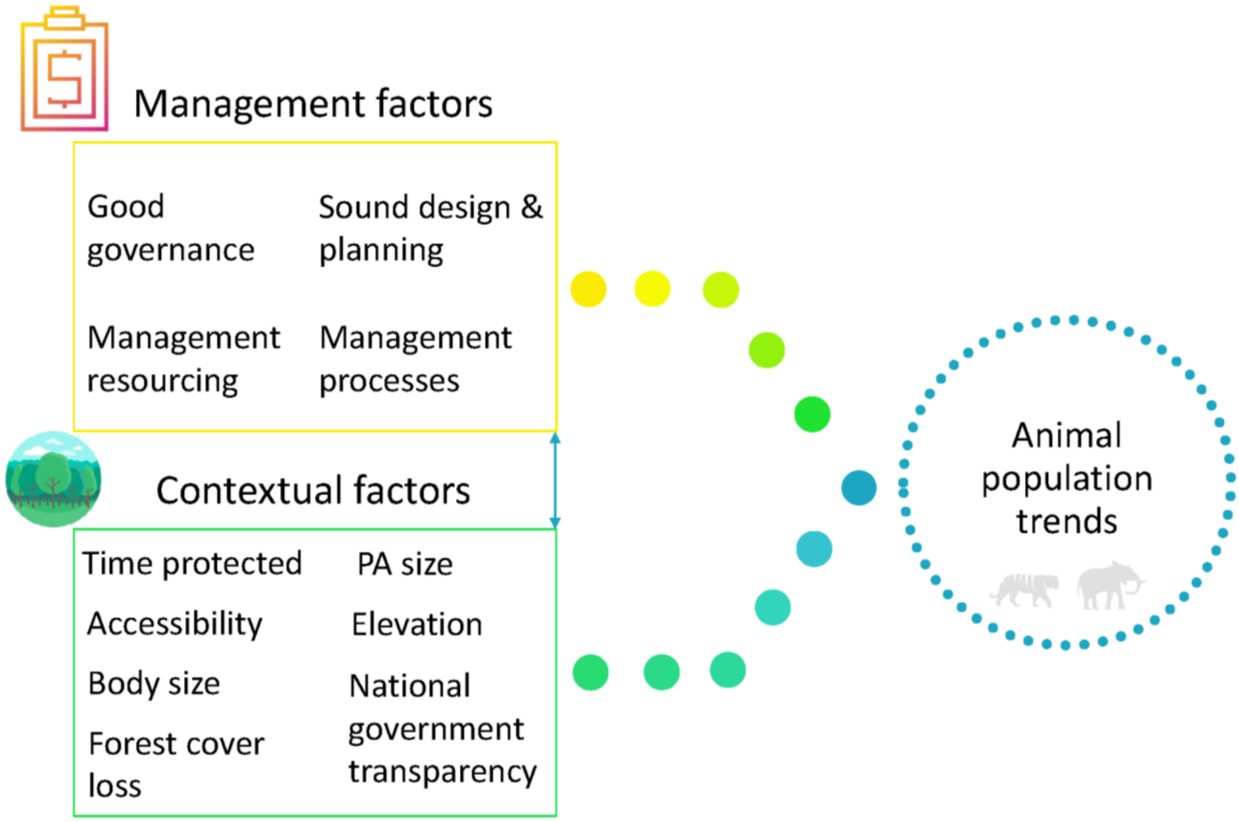


**Figure 1.** Conceptual diagram of the variables considered in our statistical modelling approach. The annual rate of change in populations over time (animal population trends) in protected areas were the dependent variables. Four management factors based on the IUCN GreenList categories: (1) good governance; (2) sound design and planning; (3) management resourcing; and (4) management processes; and seven contextual factors: (1) time protected; (2) protected area size; (3) accessibility; (4) elevation; (5) body size; (6) national government transparency; and (7) forest cover loss were the independent variables. Management and contextual factors can interact with each other (e.g. national government transparency may influence management governance at the protected area site-level) represented by the joining arrow. Details of data sources are in S1. Icons made by FreePik from www.flaticon.com and Vectortown from [www.iconfinder.com](www.iconfinder.com)

**Table 1.** Model input variables

| Dependent variable | Independent variable | | Theory of change (predicted direction of relationship) | Data source |
|---|---|---|---|---|
| *(1)* ***Animal population trends*** | Sound design and planning; good governance;) management resourcing; and management processes | | Well designed and planned, equitably governed, and effectively resourced and managed PAs have higher animal population growth (positive) | Stolton *et al.* 2007 |
| | Time protected | | Longer term protection allows for populations to recover or stabilize (positive) | UNEP-WCMC & IUCN (2018) |
| | Accessibility | | Remote areas are protected de facto (positive) | Weiss *et al.* (2018) |
| | Elevation ($log_2$) | | Higher elevation areas are protected de facto (positive) | JAXA (2018) |
| | Protected area size ($log_2$) | | Larger protected areas support viable populations (positive) | UNEP-WCMC & IUCN (2018) |
| | Government transparency | Contextual factors | Government transparency → reduces wildlife crime & illegal behaviours associated with corruption → increasing populations (positive) | Transparency International (2018) |
| | Body size ($log_2$) | | Larger species are more threatened by poaching and illegal harvesting and their populations are slower to recover due to low fecundity (negative) | Payne 2009, Jones *et al.* 2009 |
| | Forest cover loss | | Habitat loss causes animal populations to decline (negative) | Hansen *et al.* (2013) |

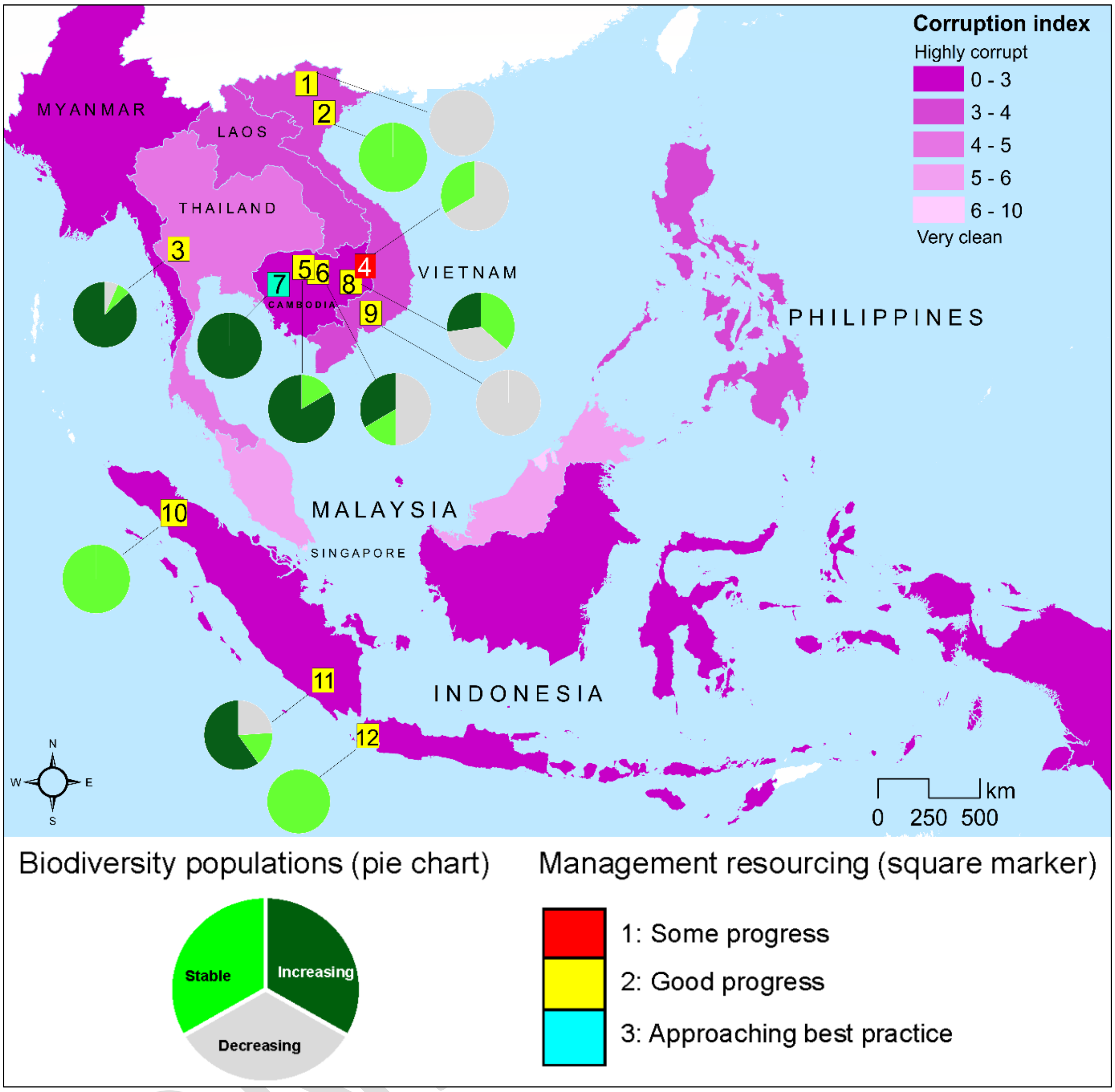


**Figure 2**. The average annual rate of change over time for all animal populations monitored (biodiversity populations) in each of the 12 protected areas assessed in Southeast Asia. The percent of animal populations that are decreasing, stable, or increasing are shown in the pie charts for each site. Countries are colour coded by the Transparency International *Corruption Perception Index* from highly corrupt (0) to very clean (10). The colour of the square marker represents the management resourcing score from 1 to 3 (1: some progress; 3: approaching best practice). Names of protected areas are: 1 = Na Hang Nature Reserve, 2 = Xuan Thuy National Parl, 3 = Thungyai Naresuan Wildlife Sanctuary, 4 = Phnom Prich Wildlife Sanctuary, 5 = Kulen Promtep Wildlife Sanctuary, 6 = Chhaep Wildlife Sanctuary (formerly Preah Vihear Protected Forest), 7 = Prek Toal Multiple Use Management Area, 8 = Keo Seima Wildlife Sanctuary, 9 = Cat Tien National Park, 10 = Gunung Leuser National Park, 11 = Bukit Barisan Selatan National Park, 12 = Ujung Kulon National Park.

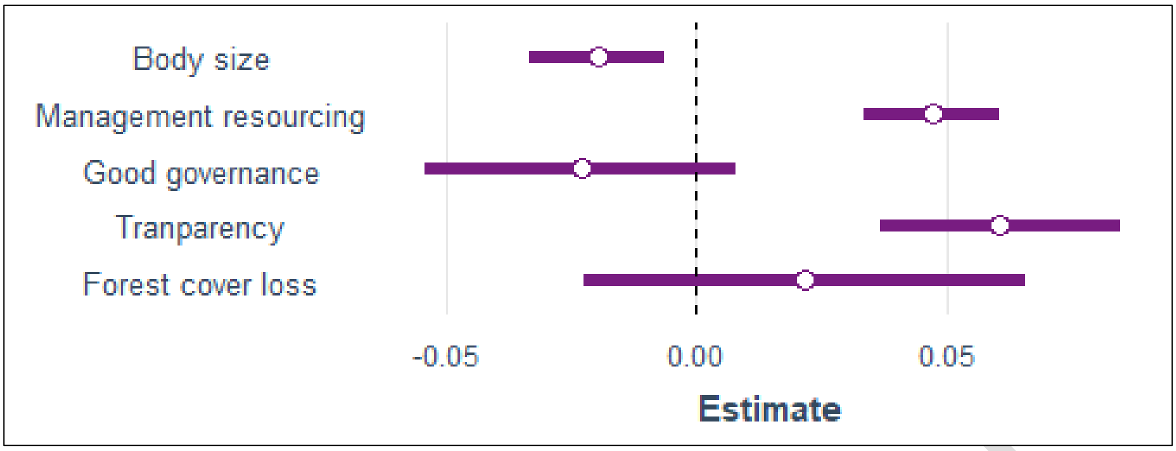


**Figure 3.** Regression coefficient estimates (scaled) of the model input variables that were included in the best-fit linear regression models based on Akaike information criterion for predicting animal population trends. Error bars are for a 95% confidence interval. Management resourcing includes budget, staff, equipment, objective setting and implementing a management and day-to-day work plan. Staff training and budget management were highly correlated with these variables and therefore omitted from the model.