Biological Monitoring for the Integrated Ecosystem and Wildlife Management Project

A Baseline Survey of Ungulate Distribution and Abundance in the Bolikhamxay Landscape, Lao PDR.

Final Report

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1 Conservation Context

1.1 Biodiversity Values in Bolikhamxay

There are two distinct ecological areas – east and west – in Bolikhamxay Province. The Annamite Mountains that run along the eastern half of the province are blanketed by a unique evergreen wet forest that is home to a high number of endemic species. This distinct habitat is found only along the Annamite Mountains of Laos and Vietnam. Much of this forest remains relatively unsurveyed. A number of new mammal species have been discovered there in recent years, such as the Saola, the Large Antlered Muntjac and the Annamite Striped Rabbit. Most of the endemic wildlife is under severe threat. A large, contiguous area of wet evergreen forest is found within Bolikhamxay Province; and is included inside a number of Protected Areas (the Nam Chouan Protection Forest, the Nam Chat/Nam Pan Watershed Protection Forest, the Phou Chom Voy Provincial Protected Area, the Phou Sithone Endangered Species Conservation Area, and the Nam Theu n Extension Biodiversity Conservation Area). In addition to unique Annamite endemics, these forests of eastern Bolikhamxay are also home to several endangered primates including gibbons, douc langurs and leaf monkeys. Due to their uniqueness, these forests are globally irreplaceable and an ecological priority for the Laos national protected area system. Due to their size and contiguity, they also hold the best chance for long-term viability of the endemic wildlife.

These eastern forests are connected through various corridors to the mixed semi-tropical forests of western Bolikhamxay. These forests are largely of a type that was historically found widely in Indochina, harboring species such as elephant and clouded leopard. Much of this habitat has now been converted for agriculture, logging and plantations. However, several protected areas, the Nam Kading National Protected Area in particular, have largely escaped this conversion and thus are of national and regional importance. The Nam Kading National Protected Area is over 1,600 square kilometers, and can likely support viable populations of many tropical mammals of conservation concern. This area is known to hold at least four Critically Endangered and Endangered primate species, a small population of Endangered Asian Elephant, a variety of small and medium carnivores, and multiple ungulate species.

1.2 Threats to Biodiversity

There are two broadly defined threats to the biodiversity of all of Bolikhamxay Province – hunting and habitat loss and modification. Hunting threatens nearly all taxa; species with larger body sizes are under particular threat. Both local residents and outsiders hunt. Some local residents are dependent on wildlife to fulfill a large proportion of their protein needs. This is particularly true of the rural residents of Bolikhamxay, who comprise 80% of the population. Local residents also hunt wildlife to sell to both national and international traders. Southern China and Vietnam have a long tradition of consuming wildlife as a luxury item and the growing middle class of these countries has led to a dramatic surge in the demand for wildlife, much of it coming from the forests of Lao PDR.

Logging and the attendant habitat degradation began in Bolikhamxay Province in the late 1980's with the overharvest of a few especially high value species. More recently economic land concessions for agroforestry plantations have proliferated across the landscape, often in the same areas that were logged in the 1980's. Habitat loss and degradation has occurred both outside and inside protected areas, particularly in Provincial Protected Areas, due to little or no on-the-ground law enforcement. In addition there is an ongoing lack of clarity and general confusion over the exact boundaries of all protected areas, which contributes to encroachment.

An underlying indirect threat that contributes to all aspects of biodiversity loss in Laos is the low technical capacity of protected area staff and lack of resources to do the needed work. Laos is still recovering from decades of conflict and consequently the education system in the country is still well below international standards.

1.3 IEWMP

The Integrated Ecosystem and Wildlife Management Project (IEWMP) is a cooperative project between the Bolikhamxay Provincial Office of Natural Resources and Environment (PONRE), the Department of Forest Resources Management (DFRM), and the Wildlife Conservation Society (WCS). The project is now in its eight year and is underpinned by a formal Memorandum of Understanding (MoU) between WCS and the Government of Lao PDR (GoL).

The objective of this project is to work in the wider Bolikhamxay landscape with a focus on protected area management and the protection of species of high conservation value, as well as on the human landscape around protected areas. The project has seen the successful implementation of a range of activities involving local communities, GoL institutions, and industry partners working in this landscape. These activities include;

- Land use zoning and demarcation
- Community engagement – village development
- Conservation outreach and awareness
- Law enforcement
- Capacity development
- Biodiversity monitoring

The project currently focuses efforts in three Protected Areas within Bolikhamxay Province. Phou Si Thone Endangered Species Conservation (PST ESC) area in eastern part of the province, the Nam Kading National Protected Area (NKD PA) in the west, and the Nam Gnouang South Protection Forest (NGS PF). The NGS PF plays an important strategic corridor role in eastern Bolikhamxay by linking three conservation forests including Phou Chom Voy Provincial Protected Area (PCV PPA) and the IEWMP is currently engaging with a private sector partner at this site.

1.4 Biodiversity Monitoring Within IEWMP

In 2007, WCS provided support to IEWMP to develop a monitoring plan for NKD PA, based on the landscape species approach (Sanderson et al. 2002). Six landscape species were chosen to represent the site (tiger, gibbon, elephant, wild pig, serow and greater hornbill) (Stindberg et al. 2007). The plan was comprehensive and ambitious, and it included multiple methods as no one technique can satisfactorily monitor an entire suite of landscape species.

A subset of the planned monitoring activities was initially carried out, but the plan was not implemented in its entirety due to constraints on funding and human resources. Although considerable effort was expended, it was not sufficient to obtain the requisite data for the extremely low density species concerned. Due to sparse data it was impossible to generate precise population estimates and as a consequence the results cannot be easily interpreted or clearly linked to management effectiveness.

Given this situation, it was decided to develop a new biological monitoring approach for the Bolikhamxay landscape. This new approach was predicated on the need to balance the requirement for scientifically rigorous data that is useful from a management perspective with practical feasibility in terms of field implementation. In 2014, WCS and IEWMP amalgamated funding from multiple donors (LifeWeb, THPC, AFD, Cargill and EU) in order to develop and implement this new approach. The survey was to be conducted across three Protected Areas in the province, Nam Kading National Protected Area (NKD NPA), Nam Nguang South National Protection Forest (NGS NPF) and Phousithone Endangered Species Conservation Area (PST ESCA).

2 Methodological Justification

2.1 Conservation Monitoring for Management

Monitoring here is defined as the process of periodically collecting information about specific ecological relevant variables within a system of interest, such as a protected area or landscape. Monitoring information is used to characterize the status of the area at different points in time for the purpose of assessing the state and drawing inferences about changes in state over time (Yoccoz et al., 2001). Conservation monitoring is crucially important for forewarning of impending species declines and extinctions, for creating triggers for management interventions, for assessing the effectiveness of management actions designed to preserve biodiversity (Lindenmayer et al., 2012).

Monitoring for conservation management purposes is most useful when it provides information for decision-making (Nichols and Williams, 2006). Monitoring that does not provide relevant information for decision-making is inefficient because it uses human and financial resources that could be directed elsewhere. Although there is sometimes a perception that monitoring which provides *any* additional information about a system is worthwhile, this approach has been heavily criticised (Legg and Nagy, 2006; Yoccoz et al., 2001) and a more useful and cost-effective approach is to clearly specify the objectives of a monitoring program. These objectives must closely reflect the stated management goals for a given area.

The overall management goal for this landscape is to conserve the globally important biodiversity of Bolikhamxay (REF). In order to assess progress towards this goal the biodiversity monitoring approach must be able to measure changes in the status of key populations over time. This requires methods which provide reliable data on species distribution, abundance or other relevant biological parameters for these populations (Nichols and Williams, 2006). However, measuring change in biological populations is often a long-term effort. Furthermore, such changes will only occur if negative pressures on populations, such as hunting, are reduced or eliminated through management interventions. Monitoring threats can therefore provide a practicable means of measuring management effectiveness in the short to medium term. Such intermediate measures of effectiveness are particularly valuable in highly dynamic landscapes such as the one concerned, and they can also provide an opportunity for interventions to be managed more adaptively (Kapos et al., 2008; Salafsky et al., 2002). Thus, monitoring should be conducted on multiple levels; for both threats and for wildlife populations.

2.2 General Sampling Framework

The design of any biodiversity monitoring strategy must consider two issues; appropriate spatial sampling and accounting for imperfect detection (Williams et al., 2002; Yoccoz et al., 2001). The issue of spatial sampling is important because areas of interest are frequently too large to be surveyed in their entirety and so a sample of smaller areas must be selected in which to focus survey effort. However, selection of sample locations must be conducted in a manner that permits inference about the entire area of interest. The issue of imperfect detectability relates to the fact most survey methods exclude the possibility that investigators will observe every animal (or sign etc.) within a given sample location and therefore the counts obtained represent an unknown fraction of the animals present in the sampled area (Pollock et al., 2002; Yoccoz et al., 2001).

Ensuring that the spatial sampling is representative can be dealt with through the use of a probabilistic sampling scheme, i.e. randomised sampling or systematic sampling with a random start (Thompson et al., 1998; Williams et al., 2002). An array of approaches have been developed to deal with imperfect detectability, such as distance sampling, capture-recapture and, more recently, occupancy based methods (Buckland et al., 2001; Mackenzie, 2006; Williams et al., 2002). These approaches can used to obtain an unbiased estimate of any state variable of interest; i.e. population density/abundance or occupancy.

Approaches which account for imperfect detection are absolute methods, whereas those that do not are classed as relative or index-based methods. Absolute methods are robust and generate more reliable information for management but they are also more technically complex and generally require a far higher level of investment to implement (Danielsen et al., 2003; Pollock et al., 2002). Index-based methods are more economically in terms of data collection but they vulnerable to numerous sources of bias and must rely on standardization to minimise this (McComb et al., 2010; Yoccoz et al., 2001). The results from index-based methods allow for more limited inference and must be interpreted with caution from a management perspective (McComb et al., 2010).

An additional consideration is the statistical power of the sampling design, that is, the ability of a monitoring approach to detect a real effect or response when it exists. Power increases with increasing sample size, and increasing size of the effect or response. Power decreases as the variance and standard error increases. The goal is to be able to develop a monitoring approach that will detect a response with sufficient sensitivity to base management decisions upon. Low power in a monitoring program means high uncertainty in interpreting the data.

Monitoring approaches which permit the estimation of detection probability, and which yield high statistical power can be costly. However, failure to take into account these considerations can result in a monitoring program which cannot reliably identify or track the impact of the management interventions.

2.3 Considerations for Monitoring in Lao PDR

Populations of large mammals within protected areas in Lao PDR are typically extremely depressed due to historically high levels of hunting pressure. These very low densities raise challenges for the design of monitoring programs as they exacerbate the difficulties associated with both spatial sampling and detectability. In addition, small sample sizes (due to a lack of detections or observations) often result in population estimates with low precision which makes reliable trend identification problematic and limits the inference that can be made from the data. This in turn complicates any assessment of how populations are responding to identified threats and subsequent management interventions.

Multiple practical constraints also apply in Lao PDR. Access to manpower, equipment and other resources is often limited and technical capacity is low. Furthermore, the extremely rugged, inaccessible terrain which characterizes many PAs further hampers the implementation of ecological surveys and greatly increases the financial and logistical costs associated with monitoring programs.

When developing approaches to monitor biodiversity in Lao, it is necessary to consider all of the above factors in order to ensure the approach is both feasible to implement and scientifically defensible. Complex monitoring approaches that monitor a large suite of wildlife species using a variety of methods might be scientifically sound, however, they may also be prohibitively expensive and exceed the capacities of national management units to implement. On the other hand, too simplistic an approach may produce unreliable or uninformative results which are not useful for management purposes.

Opportunities to employ traditional methods of population estimation in this context are limited. The implementation of capture-recapture methods is precluded by an inability to identify individuals within a population for most of the species of conservation concern and by the large spatial scale of the project. Distance sampling is not possible due to the rugged terrain and to the realistically unachievable amount of effort that would be required to obtain sufficient sample sizes (i.e. a minimum of 60-80 observations of each target species). In terms of absolute methods this leaves only occupancy-based methods to consider.

2.4 Occupancy Methods and Hierarchical Models

Over the last decade the distribution or occurrence of a population has been increasingly adopted as an alternative parameter of interest to wildlife managers. Occupancy is defined as the probability that a sampling unit is occupied by a species, or generalized to mean the proportion of an area occupied by a species (MacKenzie et al., 2002, 2006). This has been found to be a relevant and useful measure when assessing the impact of management actions, especially in long-term monitoring programs, and a large body of research now exists surrounding occupancy-type approaches (Bailey et al., 2014).

Occupancy approaches provide a means of clearly distinguishing 'true absence' of target species from 'non-detection' through the explicit estimation of a detection probability associated with each sample unit. This is done by carrying out replicated surveys across a network of grid cells or habitat patches. The approach is somewhat similar to traditional capture-recapture surveys in that "capture histories" are produced through this repeated sampling and are then used to estimate a detection probability. However, whereas in capture-recapture surveys these capture histories relate to a specific, identifiable individual, in occupancy surveys they relate to the presence or absence of *any* individual of that species in a sample unit.

Occupancy approaches can be contextualized within a more general framework of hierarchical models, both conceptually and analytically. Hierarchical models contain two explicit component models that describe variation in the data due to 1) spatial or temporal variation in the underlying ecological process and 2) imperfect observation of this process (Royle and Dorazio 2008). Within a hierarchical modelling framework either a maximum likelihood-based or Bayesian approach is employed to simultaneously estimate both the state variable of interest (i.e. occurrence or abundance) and detection probability. This kind of framework also readily allows for the incorporation of covariates hypothesized to influence either occurrence/abundance or detection probability. The theory underpinning occupancy approaches is described in detail in Mackenzie et al. (2002; 2006).

- $y_i = \{0, 1\}$ observations of presence/absence at site i
- $z_i = \{0, 1\}$ state-variable true presence or absence

Observation model:

$$y_i | z_i \sim \text{Bernoulli}(z_i p)$$

$p =$ probability of detecting species *given that it is present*

Process model:

$$z_i \sim \text{Bernoulli}(\psi_i)$$

$$logit(\psi_i) = \alpha + \beta x_i$$

Box 1 Example of hierarchical model of species occurrence.

2.5 Occupancy Surveys in Bolikhamxay

The development of general occupancy-based estimation methods, described above (MacKenzie, et al. 2006, 2002), has been further adapted to specifically fit into a Southeast Asian context (i.e. extremely low underlying densities, difficult terrain, limited access and local capacity). Under a previous regional initiative Tigers Forever (WCS/Panthera) this tailored approach has been tested relatively extensively in the region (i.e Gopalaswamy et al., 2012; Vongkhamheng et al., 2013). For the reasons listed below it was decided to base the new biological monitoring program across the Bolikhamxay landscape upon this approach. This is the second time this approach has been used in Lao PDR and there is considerable potential for it to be rolled out in additional sites in the future.

The main reasons behind the selection of this method are as follows;

- The ability of this approach to generate statistically rigorous estimates which can be used to detect and to reliably monitor trends.
- The fact that this approach can be implemented even in the challenging environmental conditions (in terms of terrain, access and other logistical constraints).

- The suitability of this approach for low density (artificially depressed) target populations and consequently sparse data.
- The relatively low technical capacity needed for this approach, meaning that it can realistically be implemented by the government agencies responsible for long-term management of protected areas.
- The broad spatial scale at which this approach can be implemented and the fact that it needs only to be rolled out every 3-5 years.
- The potential of this type of survey to provide employment to local people and to include graduate students as team leaders.
- The indirect protection function that this survey offers (due to the deterrent effect of monitoring teams being deployed across entire PAs).
- The potential for this approach to become the biological monitoring standard across all NPAs in Lao PDR.

The adoption of this approach will provide a means to achieve the following objectives.

- **Obtain reliable, scientifically defensible population estimates (occupancy and an index of detectability-corrected abundance) for key species.**
- **Map the distribution of threat indicators, specifically those relating to hunting and disturbance.**
- **Investigate spatial relationships between the occurrence of key populations, threats and management interventions such as anti-poaching patrol effort.**

3 Survey Design

3.1 Target Species

Wildlife densities in the Bolikhamxay landscape are severely depressed across almost all mammal and bird species. As described above, at such low densities obtaining the sample sizes necessary for precise population estimation for wide-ranging and rare landscape species is difficult. Furthermore, as landscape species are purposefully selected to collectively represent the full range of habitats and threats within a site, monitoring a full suite of landscape species necessarily requires multiple intensive and extensive survey methods. Given the constraints outlined in previous sections, and based on previous experience in NKD NPA, this type of landscape species monitoring is simply unfeasible in the current context. As a result, the development of new monitoring approach for Bolikhamxay involved an important shift in focus from landscape species to indicator species.

Indicator species can be a useful alternative means of monitoring changes in the level of a direct threat, because their numbers correspond to the degree of this specific threat and can be estimated more reliably and less expensively than landscape species. In Bolikhamxay, unsustainable hunting constitutes the primary threat to all species, across multiple taxa. In situations where hunting pressure is relatively low some species may be somewhat resilient to this threat. However, within these three PAs, as with all PAs in Lao PDR, hunting pressure is (or has been in the recent past) high enough that all medium and large mammal species have been reduced to well below carrying capacity and hence there is significant potential for recovery.

Four ungulates were chosen to act as indicator species for the revised monitoring approach across the Bolikhamxay landscape;

- Wild pig *Sus spp.*
- Sambar *Cervus unicolor*
- Serow *Capricornis milneedwardsii*
- Muntjac *Muntiacus spp.*

More than one species of muntjac and potentially of pig are likely to be present in this landscape but they are indistinguishable from signs and are treated as one for the purposes of this survey.

Amongst tropical mammals ungulates species are some of the most conspicuous and therefore relatively easiest to obtain information for. Importantly also, although ungulates are targeted heavily by hunters, small and medium ungulates populations can recover relatively quickly when hunting pressure is alleviated. This monitoring strategy is underpinned by the assumption that if management interventions are effective, hunting pressure will decrease. This in turn should produce a corresponding positive response by ungulate populations (i.e. a reduction in the rate of declines, the stabilization of populations, and eventual population increases). Any decrease in hunting pressure is further assumed to be beneficial to many other species within the landscape, including other ungulates, primates, carnivores and birds.

The data collected will consist of direct observations of any of these species and of any sign, such as tracks and dung. Several types of threat data will also be collected during the course of these surveys, including;

- Observations of snares or traps
- Direct encounters with humans
- Observations or signs of domestic livestock

3.2 Sampling Design

Occupancy surveys are highly sensitive to the spatial scale of the sampling design. The design process must take into account 1) the number of primary sample units required (i.e. grid cells), 2) the number of spatial replicates required within these sample units, and 3) the size of the sample units in relation to the predicted range size of target species. A balance must be achieved between all of these considerations so that the design is ecologically and practically appropriate but also statistically robust.

Grid cell size should be relatively large in relation to the range size of target species, but not so large that occupancy becomes 1 in most sample units (MacKenzie et al., 2006). The sample unit for this survey is a grid cell of 3.25km^2 and this is the standard size that has been employed for all other surveys of this type in South and Southeast Asia. A grid was laid across the total area (1657km^2) of the three PAs and approximately 500 cells lie within or on the boundaries of these areas. Due to the limited resources available to cover such a large area, only every second grid cell within the survey area was surveyed. Each grid cell was given a unique sequential ID and a coin toss was used to determine whether odd or even numbered grid cells would be selected. This ensured that the sampling was systematic but with a random start, which results in representative and even coverage. A total of 279 cells were selected for sampling; 30 cells in PST, 70 cells in NGS and 179 in NKD (see Figure 1).

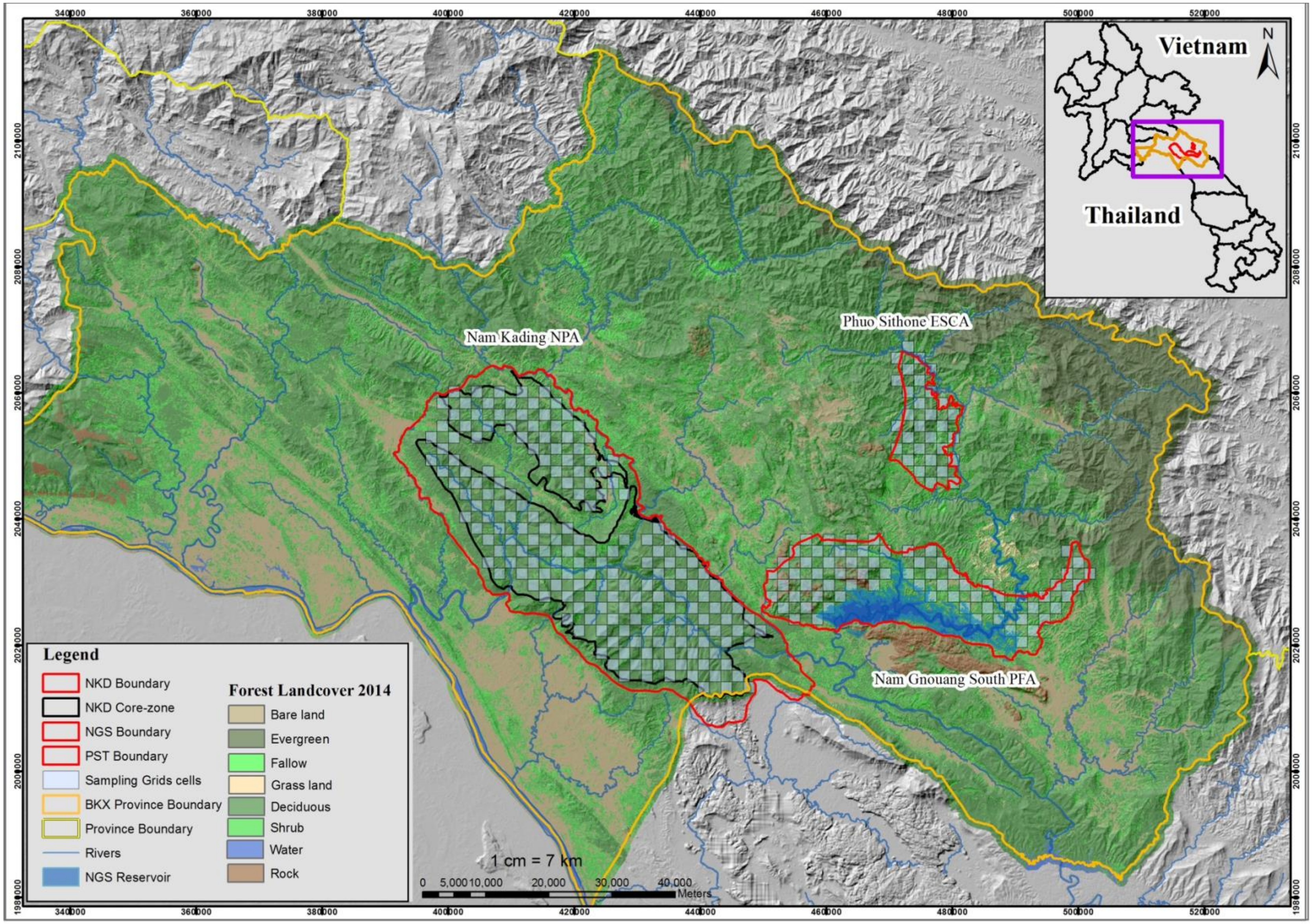

Figure 1 Sampling scheme across three Protected Areas in Bolixhamxay Province.

GIS software was used to create nine equally spaced (600 meters apart) “destination points” (points along the route to be sampled) within each grid cell. There is some flexibility allowed in sampling routes for logistical convenience (see also below). However, the route must pass through at least five destination points (see example in Figure 2) with some geometric regularity. A spatial replicate is represented by a walk of 300m, which is measured using the track log function of a global positioning system (GPS) unit. Each grid cell therefore contains nine or ten spatial replicates. Figure 3 further illustrates how such “correct” spatial replicates must be formed

Survey duration was approximately six months, and took place during the dry season in order to maximise chances of finding fresh sign, as well as for logistical ease. During the survey period cells are assumed to be closed to changes in occupancy. Although animals will actually move in and out of the sample units, these movements are assumed to be random so that the occupancy estimator remained unbiased (Mackenzie et al. 2006).

Such an extensive survey should only need to be repeated at three year intervals. Future surveys will allow for the determination of population trends, and of rates of local extinction and colonization between sampling periods using multiple season occupancy models. The survey timing and the interval between successive surveys should remain constant over time.

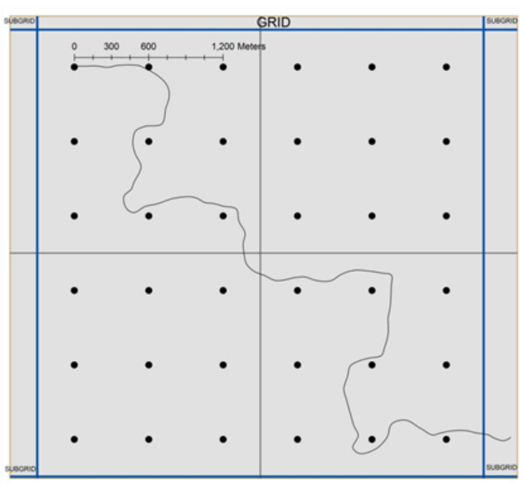

Figure 2 Four grid cells, with two surveyed.

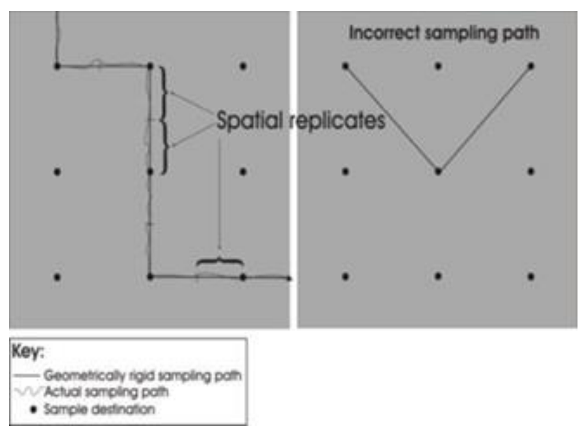

Figure 3 Examples of spatial replicates.

3.3 Data Collection Protocols

Six survey teams were established, each including a team leader assisted by a deputy team leader. Both of these individuals were experienced in animal sign identification and trained in data recording and navigation (see the Bolikhamxay Landscape Ungulate Survey Training Report for more details). They were assisted by two local village guides and four porters per team.

Teams have flexibility in choosing their sampling routes through the destination points in each grid cell, in order to facilitate ease of access etc. They must, however, pass through at least five destination points and mandatorily through the center point. Some sampling point destinations may be located in extremely rugged terrain (i.e. steep cliffs or deep gorges). In such instances terrain, the survey team must still attempt to reach within 100 m of the point destinations. The teams must attempt to walk geometrically rigid sampling paths insofar as possible between destination points (Figure 2), searching thoroughly for animals and animal signs (fresh tracks and fresh dung) and as well as humans and signs of human presence (snares, camps). The teams can deviate from a direct route between points by as much as 100m on either side of the line, if necessary, to maximize their chances of detecting sign. The search effort should cover the entire extent of habitat within the grid cell and teams are encouraged to focus their searches intensively along animal trails, rivers banks, and stream beds, and also around mineral licks and wallows.

The teams treat every 300m covered (using GPS distances) as a spatial replicate for the purposes of data recording. This requires them to mark a center point between each destination point, to mark the end of one replicate and the beginning of the following one (except for first and last destination point within a grad cell). This allows the survey teams a greater degree of flexibility than pre-marking (on the GPS units) the start and finish of spatial replicates.

Presence/absence of each sign for each species, and each threat, in a given spatial replicate is recorded by denoting a 1 if the sign is detected or a 0 if it is not. Within each replicate if just one or multiple signs of the same type and from the same species (i.e. one pig track or many pig track) are encountered this is recorded as a 1. Direct observations of both humans or animals are recorded as counts (number of individuals) per replicate. This is because it is generally not possible to determine for certain how many individuals produced a quantity of sign but for direct sightings this is observed.

One survey team member is responsible for navigation. Scale 1:50,000 topographic maps are used to plan routes and navigate through the rugged terrain (Figure 5). Grid cells and destination point locations are included on the maps supplied to teams and on the GPS units carried by teams. For logistical convenience and to deal the difficult terrain, detailed planning is made in advance by all teams.Each grid cell requires one-three days to complete, depending on terrain. The sampling area is covered in a sweep from one end to the other instead of arbitrarily sampling grid cells in a patchy or sporadic manner. This is to aid logistical efficiency and to avoid biases caused by seasonal migration of species.

4 Results

4.1 Field Implementation

Following a period of training and planning (See Appendix I: Bolikhamxay Ungulate Sign Survey Training Report) field teams were first deployed in February 2015 and surveys continued until completion in July 2015. The six field teams were initially split between PST ESCA and NGS PFA, which were completed in one and months respectively. Subsequently all teams then moved to NKD NPA which is by far the largest of the sites, but also the most accessible and familiar to the teams, hence being left until last.

All of the planned survey effort was completed in PST ESCA, but in both NKD NPA and NGS PFA, a small number of grid cells were not surveyed, or incompletely surveyed (See Figures 4, 5 & 6). This was due to severe access constraints caused by exceptionally rugged terrain, and attendant safety concerns. A total of 264 grid cells were visited by survey teams, out of a potential 279. This resulted in a total number of 2622 spatial replicates (300m segments). Ungulate presence was recorded in 1097 of these replicates. The vast majority of signs encountered were tracks (present in 1009 replicates) rather than dung (present in 186 replicates). Direct observations of pig or muntjac were recorded in 15 replicates.

For further details relating to survey implementation at each of the sites, see the respective Implementation Reports (Appendices II, III & IV).

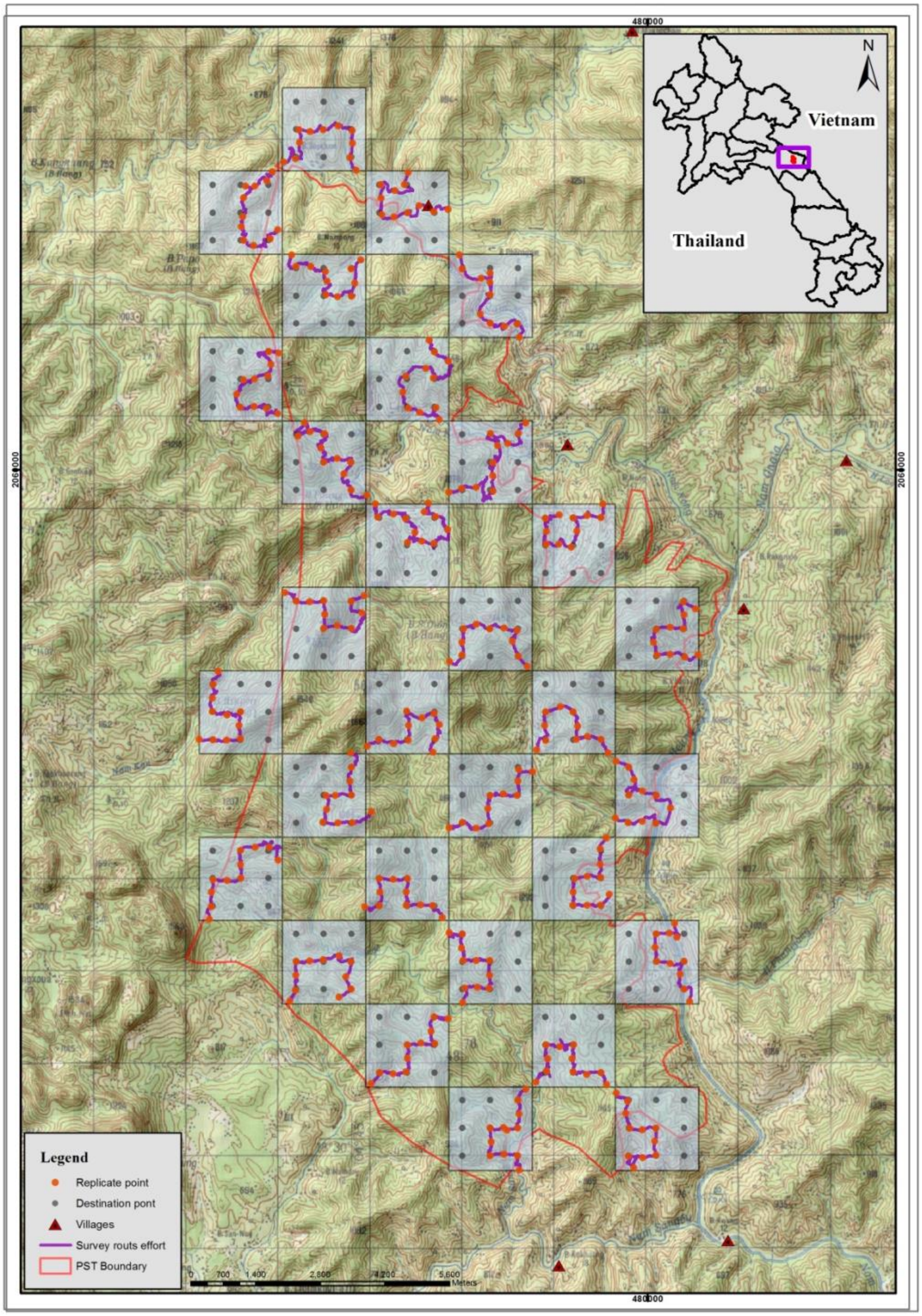

Figure 4 Sampling effort in PST ESCA.

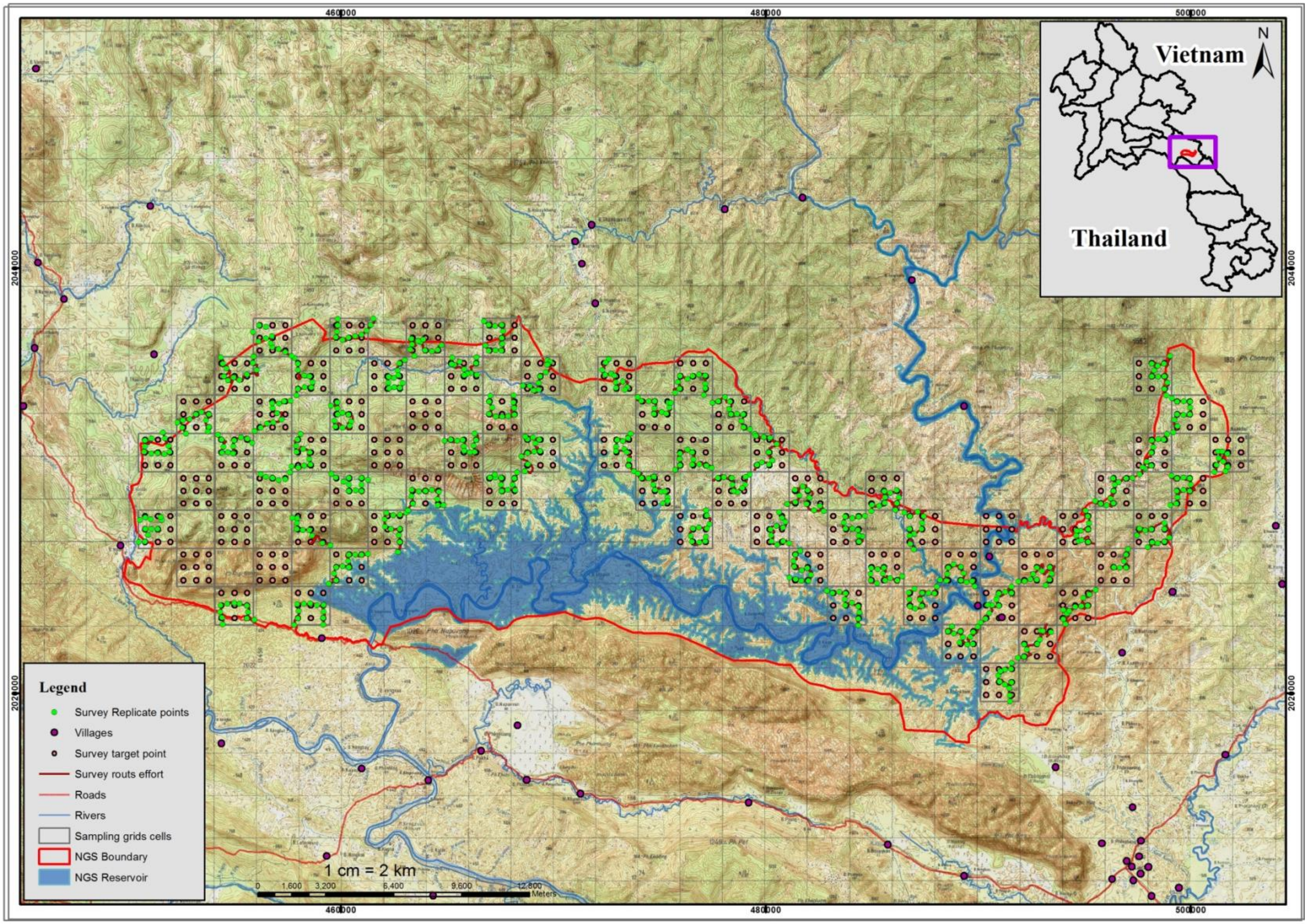

Figure 5 Sampling effort in NGS PFA

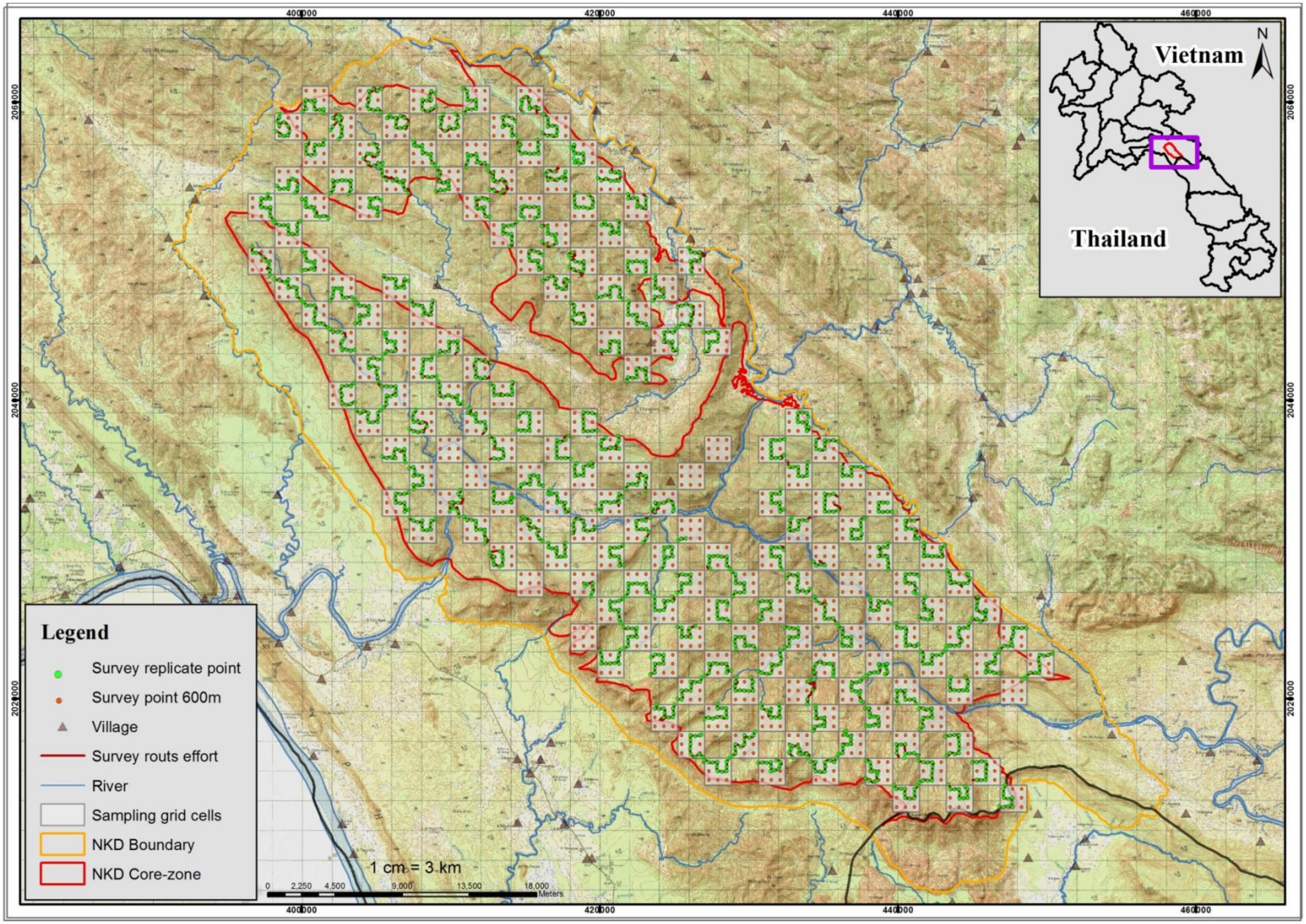

Figure 6 Sampling effort in NKD NPA

4.2 Data Analysis

Two primary statistical models were utilized in this analysis; 1) the standard single season occupancy model developed by MacKenzie et al. (2002, 2006), and a subsequent extension of that; 2) the Royle-Nichols abundance model (Royle and Nichols, 2003). A maximum likelihood approach is used for statistical inference throughout this analysis but it should be noted that these models can also be used within a Bayesian framework.

MacKenzie et al. (2002) devised a model for estimating the site occupancy probability (or proportion of area occupied: PAO) for a target species, in situations where the species is not guaranteed to be detected despite present at a site. Let ψ be the probability a site is occupied and $p[j]$ be the probability of detecting the species in the jth survey, given it is present at the site. A probabilistic argument can be used to describe the observed detection history for a site over a series of surveys. For example; the probability of observing the history 1001 (denoting the species was detected in the first and fourth surveys of the site) is:

$\psi \times p[1](1-p[2])(1-p[3])p[4]$.

The probability of never detecting the species at a site (0000) would therefore be,

$\psi \times (1-p[1])(1-p[2])(1-p[3])(1-p[4]) + (1-\psi)$,

which represents the fact that either the species was there, but was never detected, or the species was genuinely absent from the site ($1-\psi$). By combining these probabilistic statements for all N sites, maximum likelihood estimates of the model parameters can be obtained.

The model framework of MacKenzie et al. (2002) is flexible enough to cope with missing observations, i.e. occasions when sites were not surveyed. This approach also enables parameters to be a function of covariates. For example, occupancy probability may be a function of some anthropogenic or environmental conditions, while detection probability could depend upon weather or substrate. The modelling process therefore allows relationships between occupancy state and site characteristics to be investigated. Covariates are entered into the model by way of the logistic model (or logit link).

A key assumption of these models is that all parameters are constant across sites, an assumption which in many instances may not be satisfied in reality. If heterogeneity is present it can be dealt with by including relevant covariates in the analysis to account for it. If there is unmodelled heterogeneity in occupancy probabilities, estimates will still represent an average level of occupancy, provided detection probabilities are not directly related to the probability of occupancy. However, unmodelled heterogeneity in detection probabilities can cause occupancy to be underestimated.

Another major assumption of these models is that the occupancy state of the sites does not change for the duration of the surveying. Situations where this may be violated, for instance, would be for species with large home ranges, where the species may temporarily be absent from the site during the surveying. If this process of temporary absence from the site may be viewed as a random process, (e.g., the species tosses a coin to decide whether it will be present at the site today), then this assumption may be relaxed. In addition, it is important to note that the model assumes closure of the sites at the species level, not at the individual level, so there may be some movement of individuals to/from sites without overly affecting the model.

The Royle-Nichols abundance model (Royle and Nichols, 2003) was developed to estimate population size from temporally replicated presence/absence data (from point-counts or other types of surveys) at a number of sample sites. This model assumes that heterogeneity in detection probability among sites is due underlying heterogeneity in abundance (more individuals lead to higher probability of detecting the species at the site). This model can be used to generate estimates of not only occupancy but also an abundance index of animal clusters per grid cell.

Prior to analysis the detection and non-detection of target species in each grid cell was used to produce a detection history matrix for the four individual target species, for the target species combined, and for threats. A number of covariates thought to be influential on either detection probability or occurrence probability were identified and values for these covariates were obtained from existing GIS layers, or from field data collected during the course of the field survey (See Table 1).

Covariates	Detection probability	Probability of occurrence	Abundance
Elevation	–	+	+
Proximity to villages		–	–
Patrol effort		+	+
Occurrence of threats		–	–

Table 1 Predictions regarding directional effects of selected covariates on parameters of interest.

All analyses were carried using the R package “Unmarked” (Fiske and Chandler, 2011). Parameters estimated by all models included detection probability (p), probability that a site is occupied (Ψ) (leading to proportion of area occupied or used). An additional parameter relating to abundance (λ) is generated by the Royle-Nichols model (2003).

4.3 Covariate Effects

Multiple models, containing various combinations of covariates, were used to investigate a *priori* predictions regarding factors potentially affecting abundance and detectability (See Table 1). Candidate models were compared and ranked using Akaike's Information Criterion (AIC). The most parsimonious model was selected using ΔAIC, the distance in AIC units between ranked models. As a rule of thumb, it is assumed models with ΔAIC <2 are broadly equivalent in terms of fit (Burnham and Anderson, 2002). Table 2 summarizes the results of the covariate modeling process for each species and for two composite indices created by combining species (all ungulates and all ungulates except pig). A ranked list of all the candidate models for all species is provided in Appendix V.

Covariates Tested									
Species	**Replicate elevation**	**Grid cell elevation**	**Distance to village**	**Km of LE patrols**	**Cow Psi**	**Hunt Psi**	**Threat Psi**	**Snare Psi**	**Significant Covariates**
Pig	**+**	**-**	**+**	**-**	**-**	**-**	**-**	**-**	distance to village, cattle presence
Muntjac	**-**	**-**	**-**	**-**	**-**	**-**	**+**	**-**	elevation, distance to village
Serow	**+**	**+**	**-**	**+**	**-**	**+**	**-**	**+**	elevation
All ungulates	**+**	**-**	**+**	**-**	**-**	**-**	**-**	**-**	rep elevation, distance to village, km patrolled, cattle presence
Ungulates < pig	**-**	**-**	**-**	**-**	**+**	**+**	**+**	**+**	rep elevation, distance to village

Table 2 Covariate effects on ungulate detectability, occurrence and abundance. Detection probability was modelled only as function of replicate elevation. The directional effect of all covariates is indicated by symbol in each respective column and row but the covariates which were included in top-ranking models are listed in final column.

4.4 Occupancy and Abundance Estimates by Species

Top ranking models (See Appendix V) were used to produce estimates and associated confidence intervals for all model parameters. Estimates for individual species and for all species combined are presented in Table 3.

Mackenzie Occupancy Model (Probability of occurrence/proportion of sites occupied)								
Species	***p***	***p*(SE)**	**Ψ**	**95%LCI**	**95%UCI**	**PSO**	**95%LCI**	**95%UCI**
Pig	0.44	0.01	0.81	0.75	0.85	214	214	221
Muntjac	0.22	0.01	0.46	0.52	0.52	111	111	264
Serow	0.17	0.02	0.09	0.06	0.13	20	20	54
Sambar	0.09	0.08	0.01	0.00	0.05	2	2	2
Ungs <pig	0.22	0.01	0.50	0.44	0.56	120	120	264
Ungulates	0.49	0.01	0.86	0.82	0.89	225	225	227

Royle-Nichols Model (Group abundance)								
Species	***p***	***p*(SE)**	**λ**	**95%LCI**	**95%UCI**	**N groups**	**95%LCI**	**95%UCI**
Pig	0.25	0.02	1.80	1.53	2.11	438	282	913
Muntjac	0.16	0.02	0.70	0.58	0.84	137	115	465
Serow	0.15	0.03	0.11	0.07	0.16	23	21	145
Sambar	na	na	na	na	na	na	na	na
Ungs <pig	0.15	0.02	0.78	0.62	0.98	153	126	493
Ungulates	0.27	0.02	2.03	1.88	2.39	512	312	1009

Table 3 Parameter estimates for "best" models, as determined by AIC. Upper and lower confidence intervals are included for Ψ (probability of area occupied) and λ (mean abundance per grid cell). More complex models for sambar were not run due to insufficient data.

4.5 Species and Threat Prevalence Across Sites

Results yielded by top-ranking models (See Table 3 & Appendix V) were broken down by site for further comparison and are presented in Table 4. Estimates of threat occurrence were broken down by site and by indicator type and are presented in Table 5.

	Proportion of site occupied			*Mean no. of groups/grid cell*		
	PST ESCA	**NKD NPA**	**NGS PF**	**PST ESCA**	**NKD NPA**	**NGS PF**
Wild pig	83%	85%	72%	2.21	2.07	1.32
Muntjac	54%	44%	49%	0.80	0.69	0.73
Serow	18%	7%	10%	0.21	0.10	0.11
Sambar	0%	0%	4%	_	_	_
Ungulates x 3	61%	47%	54%	0.92	0.76	0.83
All ungulates	90%	87%	80%	2.59	2.26	1.60

Table 4 Estimates of POA (proportion of area occupied) and mean abundance index values.

	PST ESCA	**NKD NPA**	**NGS PF**
Cattle occurrence	53%	11%	75%
Snare occurrence	43%	27%	38%
Hunting occurrence	67%	44%	62%
Human occurrence	13%	16%	14%
All threats occurrence	73%	58%	78%

Table 5 Estimates of POA (proportion of area occupied) for threat indicators. Hunting occurrence is a combination pf snares and hunting camps. All threats includes snares, hunting camps and humans (but not cattle).

4.6 Species and Threat Distribution Maps

Top ranking models for each species, as presented in Table 3, were used to produce spatially explicit maps of abundance across the landscape for each species and spatially explicit maps of occurrence for different threat categories. These are presented in Figures 7–15.

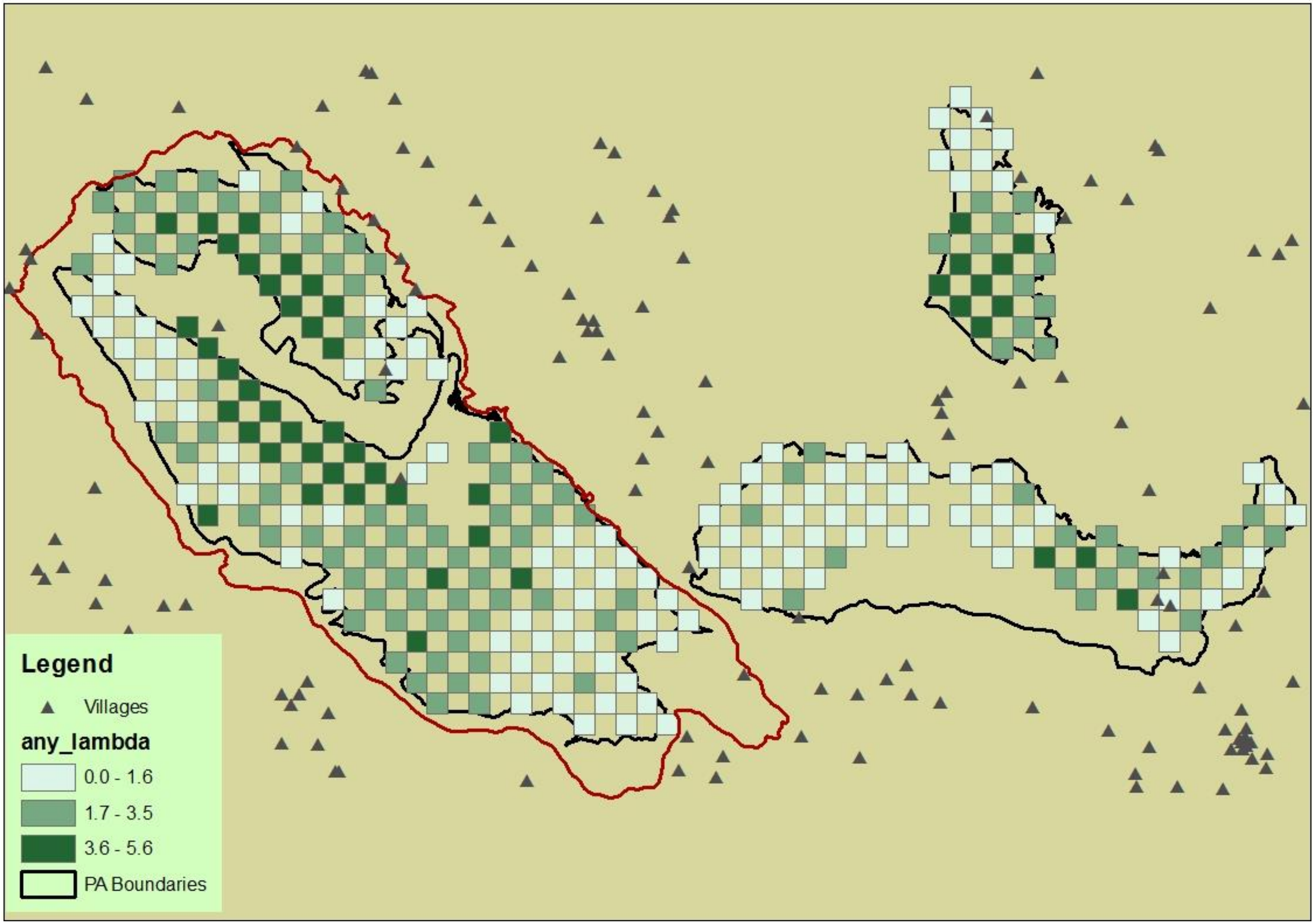

Figure 7 Abundance index for all ungulates combined.

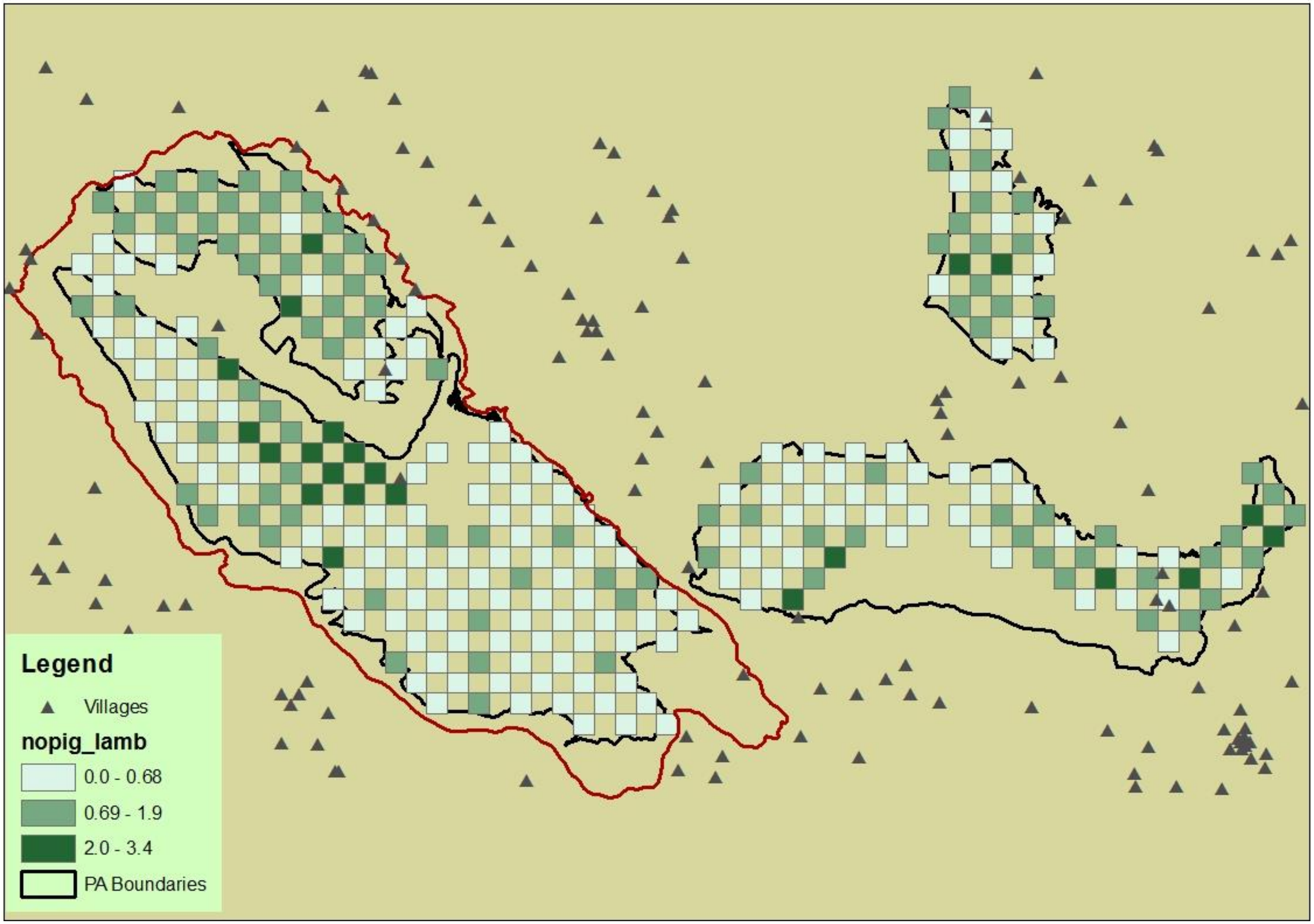

Figure 8 Abundance index for all ungulates except pig.

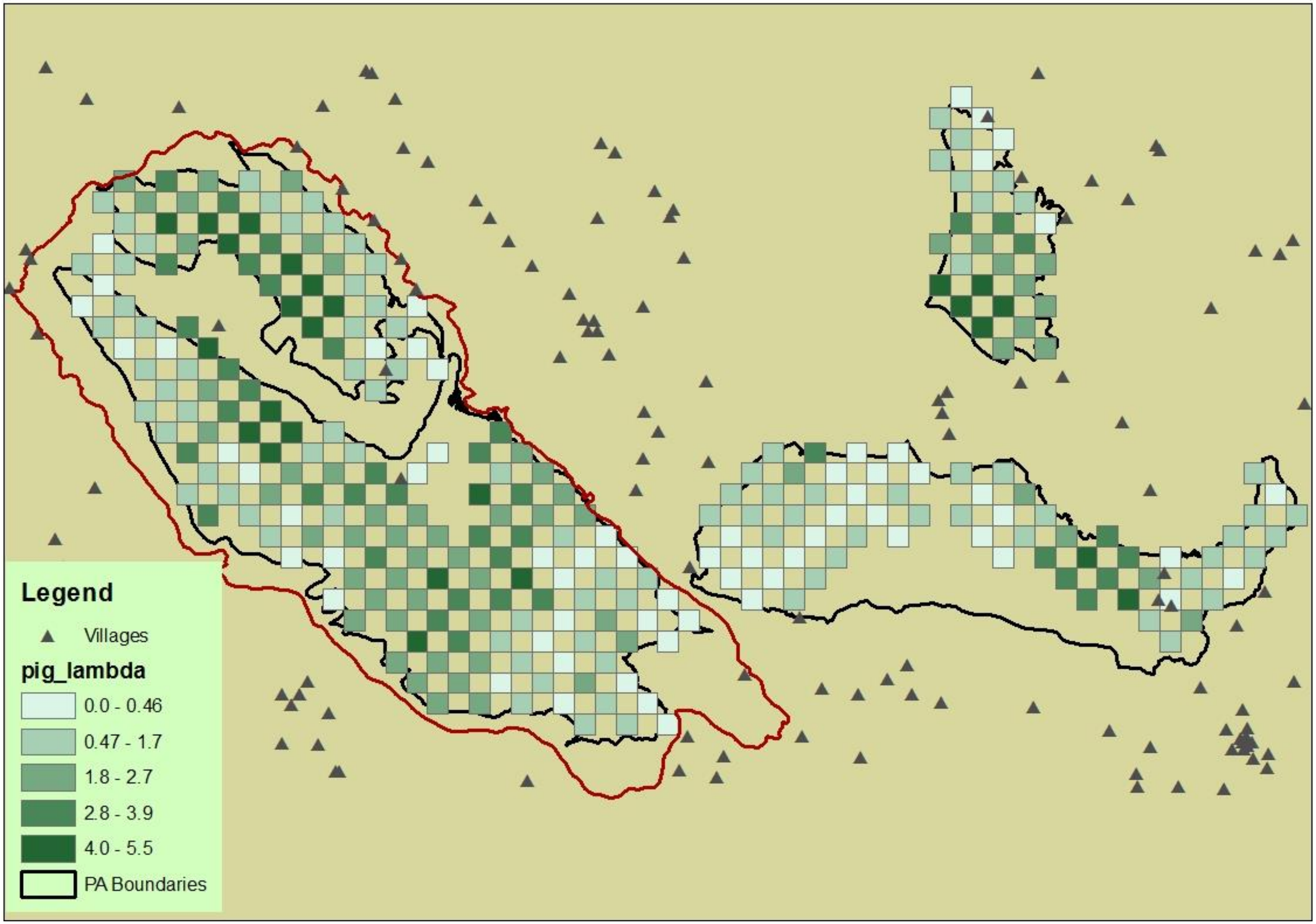

Figure 9 Abundance index for wild pig.

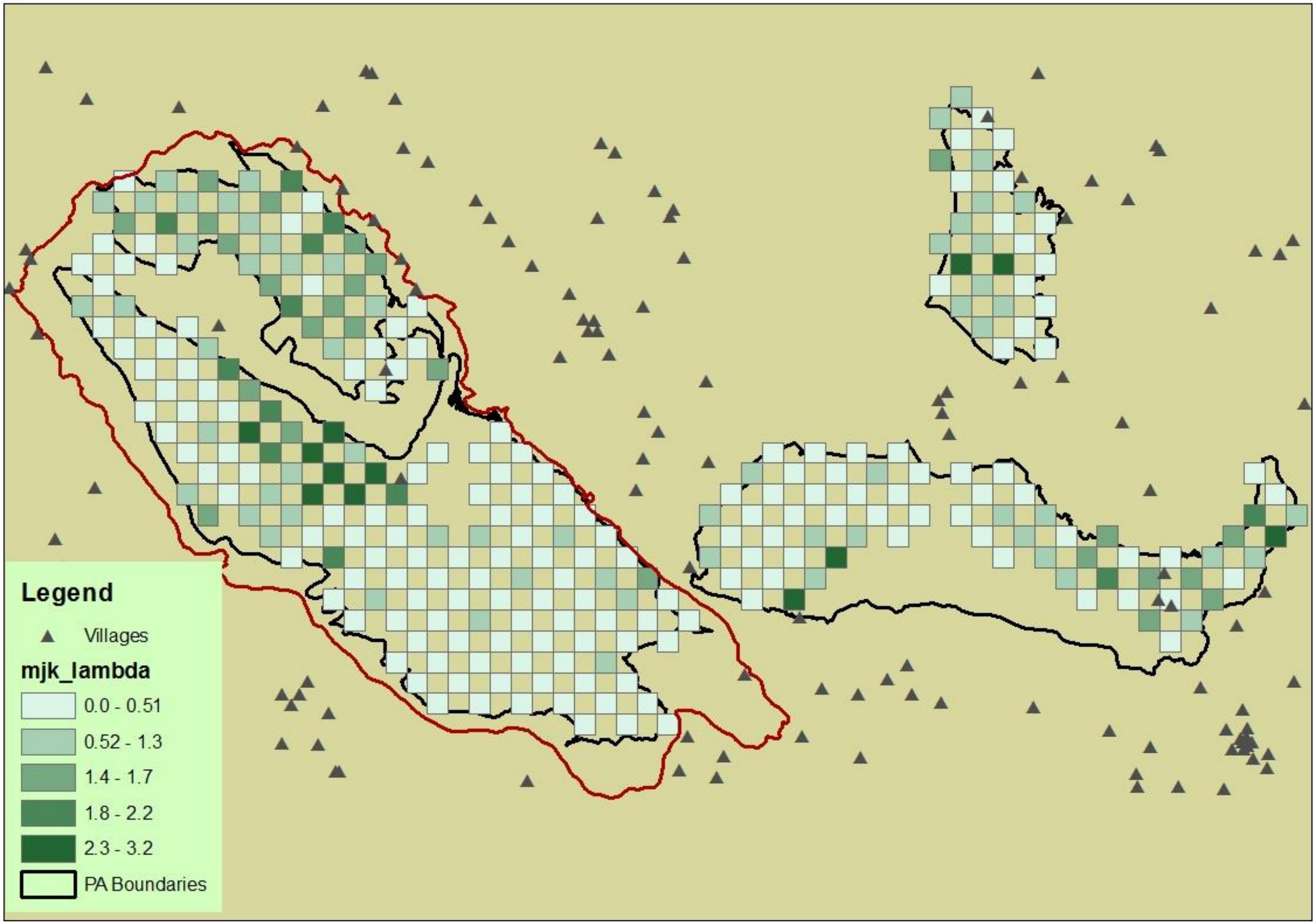

Figure 10 Abundance index for muntjac.

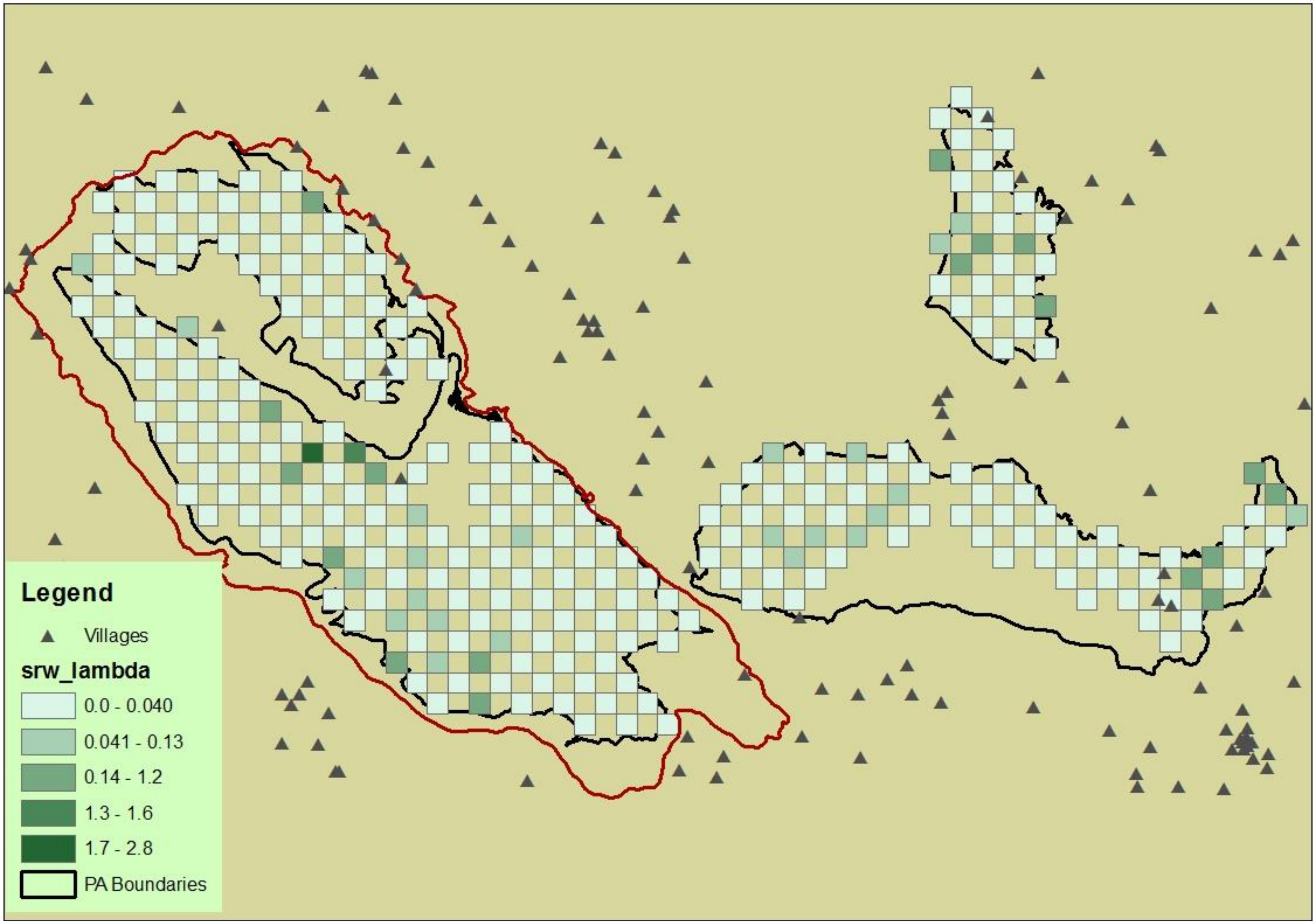

Figure 11 Abundance index for serow.

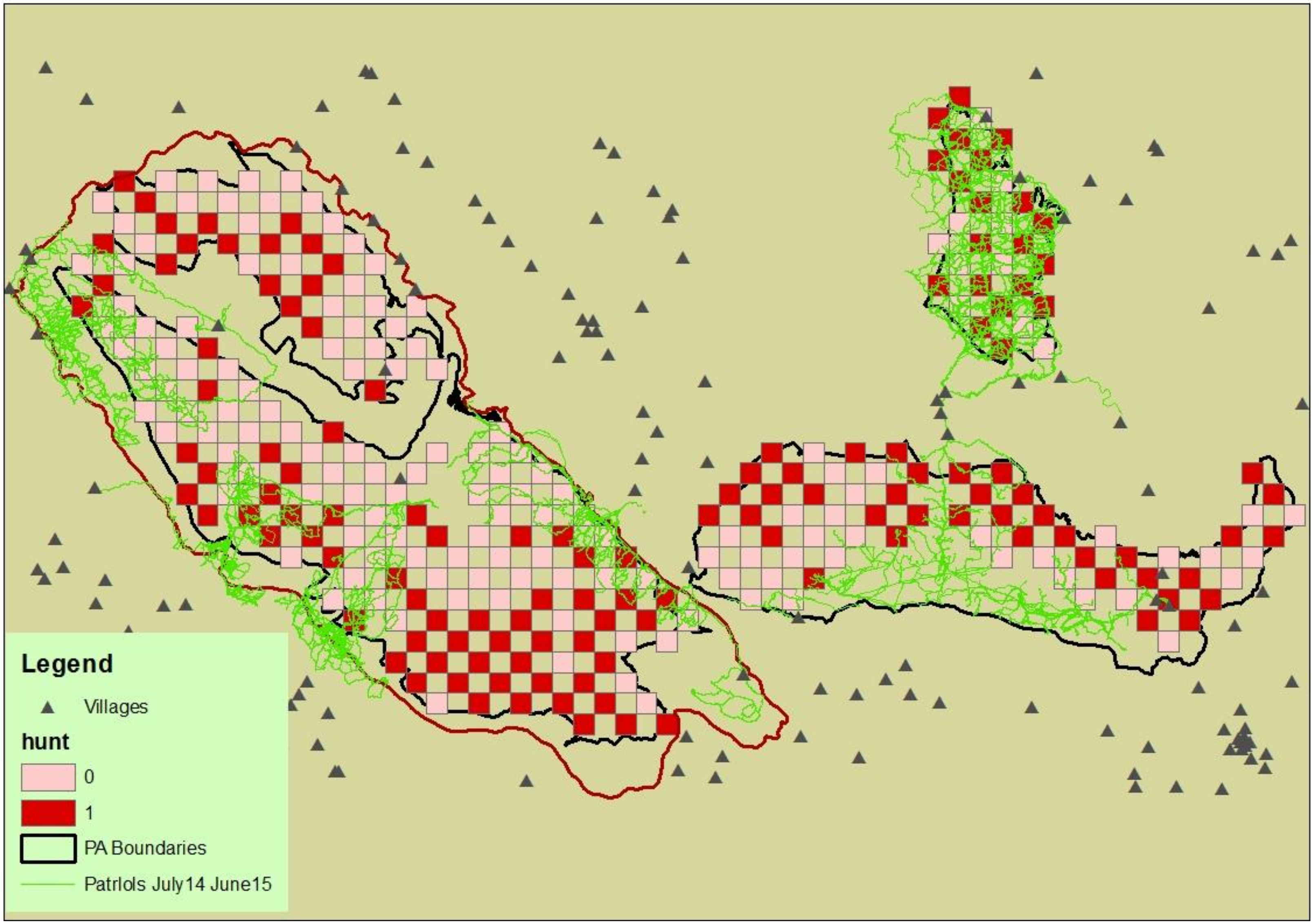

Figure 12 Probability of hunting occurrence overlaid with patrol effort during a one-year period.

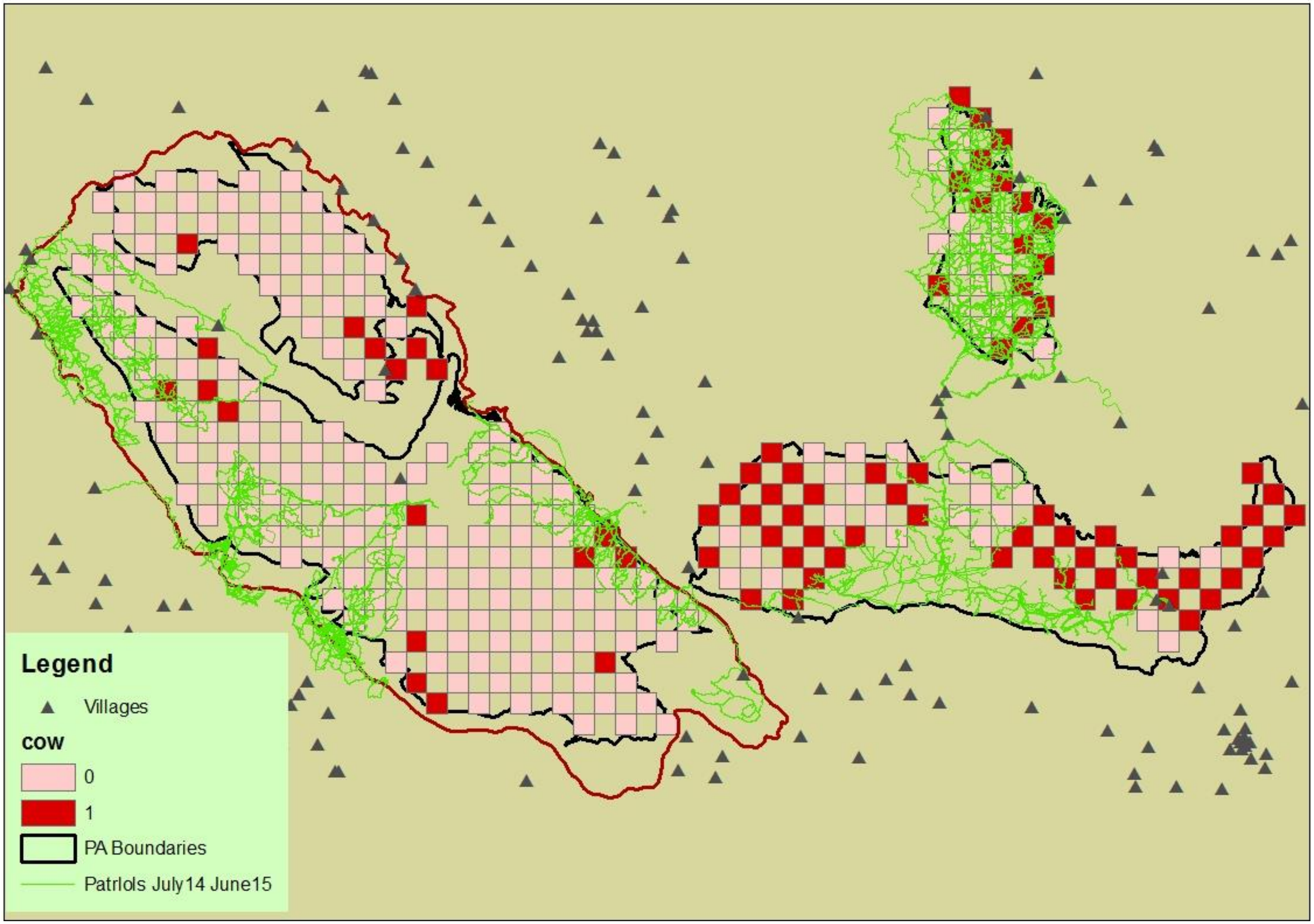

Figure 13 Probability of cattle occurrence overlaid with patrol effort during a one-year period.

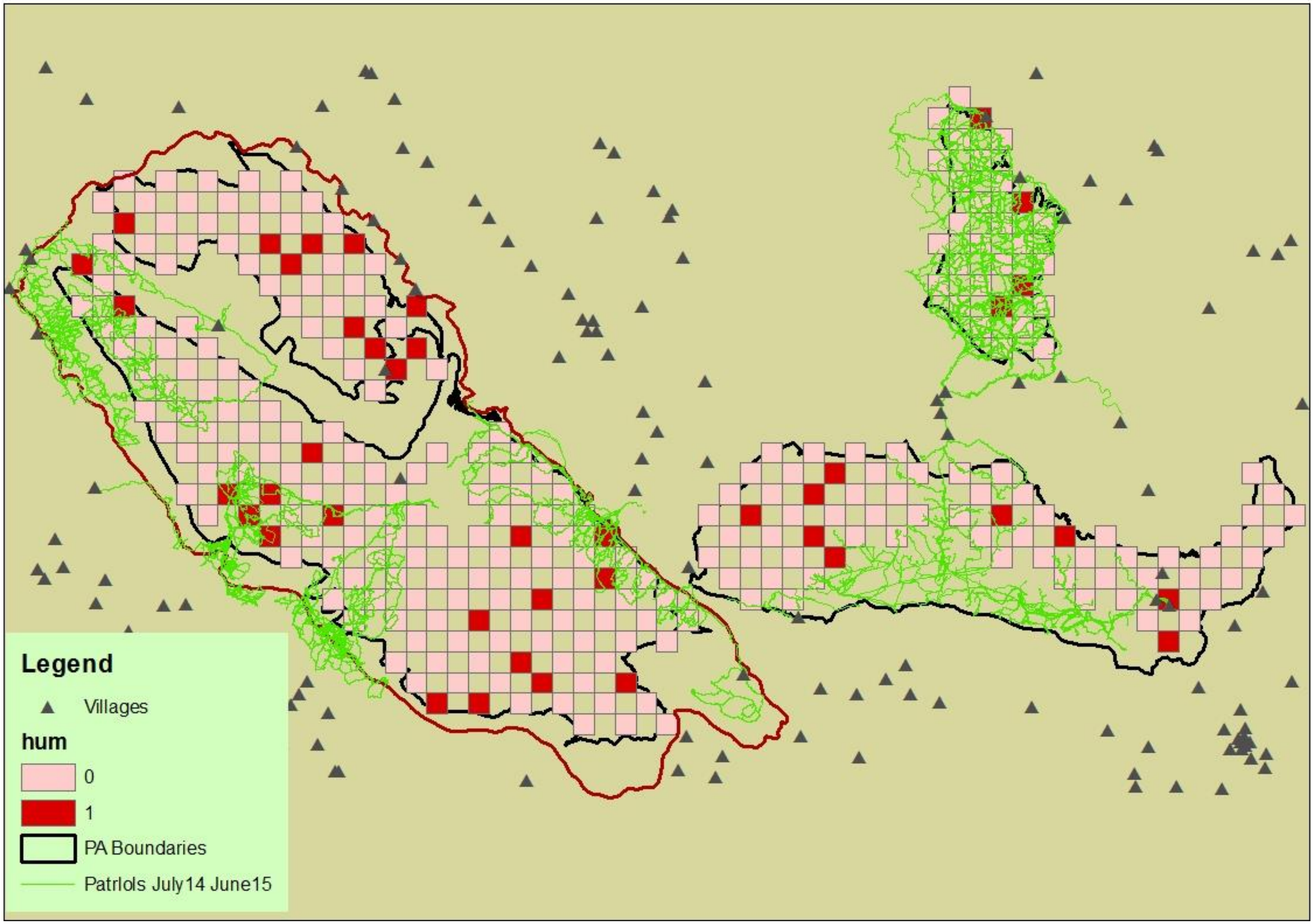

Figure 14 Probability of human occurrence overlaid with patrol effort during a one-year period.

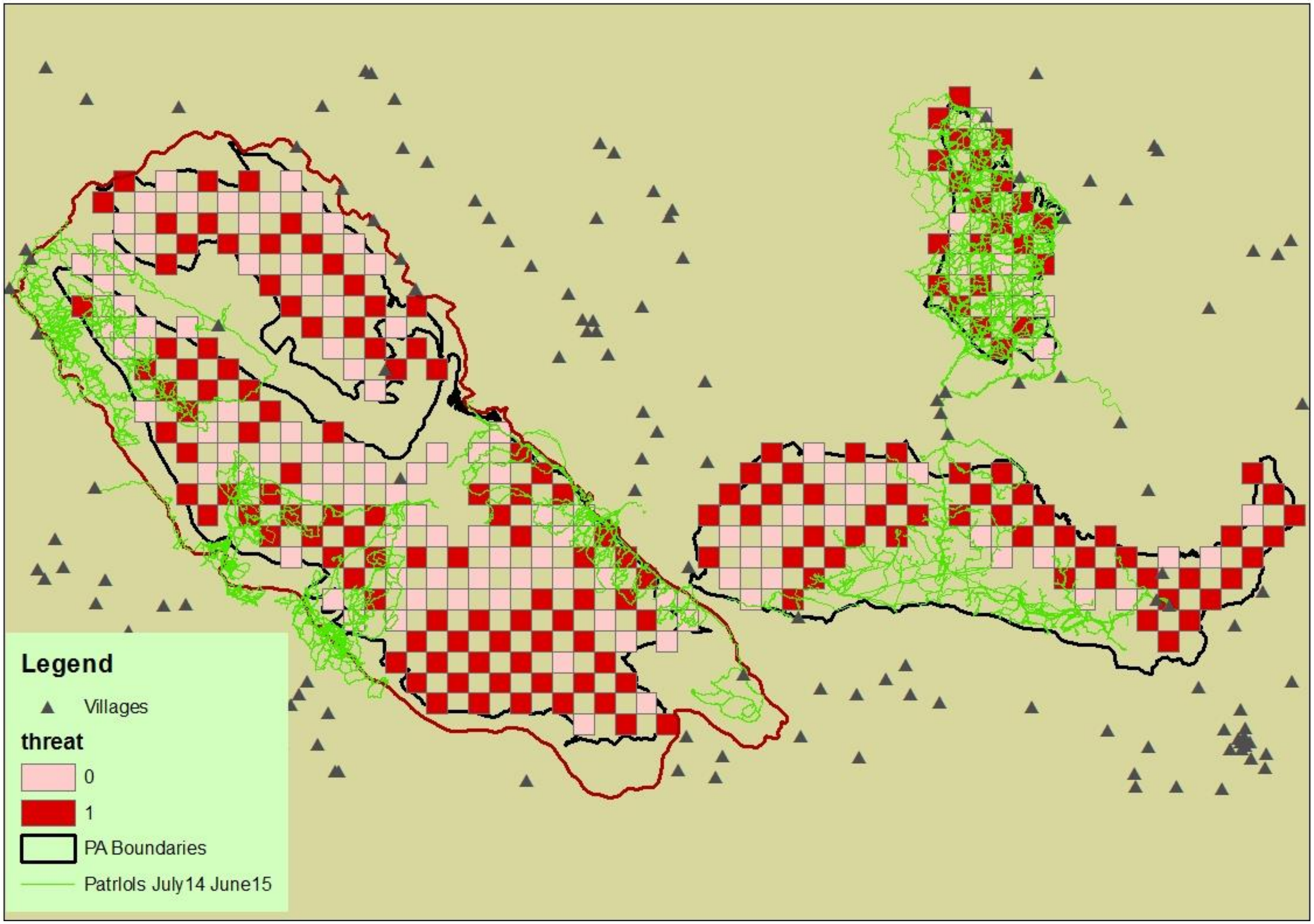

Figure 15 Probability of any threat indicator occurrence overlaid with patrol effort during a one-year period.

5 Discussion

5.1 Species Occupancy and Abundance

It is important to note that 100% accurate identification of species by signs is not possible, even by well-trained and experienced field teams (Duckworth and Hedges, 1998). Although there are obvious and consistent differences between species in a "typical" sample of dung or tracks, a proportion of signs will not be clearly identifiable, and maybe incorrectly assigned to a given species. Dung is more easily identifiable than tracks in most cases, but relatively few dungs signs were recorded during this survey. Potential mis-identifications, possibly resulting in both false negatives and false positives, is a general issue which is associated with all sign-based surveys, but it should be borne in mind when interpreting apparently species specific results.

Overall occupancy rates and abundance index values are extremely low for all species with the exception of wild pig. This indicates that underlying densities are depressed and that populations are likely to be well below the potential carrying capacity of this landscape. Estimates are lower than those obtained from a comparable 2008 survey carried out in Nam Et Phu Louey NPA in Northern Laos (Vongkhamheng et al., 2013) and seem far below densities estimated for ecologically similar areas elsewhere in tropical monsoonal Asia (Karanth et al., 2004; Karanth and Sunquist, 1992; Kawanishi and Sunquist, 2004; Srikosamatara, 1993). This is not an unexpected result, as it widely supposed that mammal populations across the entirety of Lao PDR have been drastically reduced from "natural" levels as a result of intense hunting pressure (Corlett, 2007; Harrison et al*., in press*).

This supposition is supported by the apparent differences in occupancy and abundance values across species. Wild pig populations persist at relatively high levels across all sites, followed by muntjac at medium levels. In contrast, both serow and sambar populations persist at worryingly low levels, with evidence for sambar restricted to just three replicates at only one site. Wild pig, and some muntjac species are believed to be relatively resilient to human disturbance and potentially to hunting (O'Kelly et al., 2012; Timmins et al., 2015) Sambar, on the other hand, are thought be sensitive to hunting and to disturbance, and are likely to be a preferred prey item for hunters (Timmins et al. 2015). The same may be the case for serow, which are certainly of high value for trade, but the species is habitat specific to a greater degree than the other ungulates species concerned and this will greatly influences patterns of occurrence and abundance (Duckworth et al., 2008). In fact, the inaccessibility of much of the species preferred habitat is likely to explain the apparently higher serow abundance when compared to sambar.

In general occupancy rates and abundance values were highest in PST NPA and lowest in NGS PF. These results also correspond to expectations. Although NKD has been the focus of management interventions for eight years and has received a significantly greater level of investment, it is also far larger than either of the other sites. Thus, the investment in NKD has been spread across a much greater area comparatively, and many sectors of the site receive little or no management attention. In PST, although active management efforts were initiated just three years ago, this site is small in size and thus has a far higher ratio of investment per unit area. In addition, PST is situated in a more inaccessible location, which may reduce threat levels. This interpretation is slightly complicated by the fact that PST also exhibits a slightly higher probability of hunting occurrence (both snares and camps) than the other sites. This probably reflects higher historical levels of hunting, with old snares lines and camps still being visible. NGS is somewhere between the other two sites in terms of size, but has also been the focus of management attention for a relatively short time (two years). Furthermore, the NGS site's designation as a Protected Forest affords it a lower level of legal protection than either of the other sites in this study. This is also evident in the high probability of cattle occurrence at this site.

5.2 Determinants of Occurrence and Abundance

The modelling results (Table 2) did not provide overwhelmingly strong support for any of the *a priori* predictions other than that occurrence and abundance are strongly influenced by proximity to settlements. This effect differed between species, however, with an apparent positive relationship between wild pig (and all ungulates combined) and distance to village whereas there was a negative relationship between this covariate and muntjac occurrence. This may reflect the fact that muntjac can in fact persist well in the heavily degraded habitat typically found in the vicinity of villages, whereas other species are more likely to occur further away from villages.

Elevation was also a relatively important predictor, but again this differed across species. Unsurprising there was a strong positive effect of elevation on serow occurrence, and a strong negative effect of elevation on muntjac occurrence, most likely as a result of differences in habitat preferences. Interestingly, with the exception of muntjac, elevation also had a relatively strong and positive effect on detection probability (i.e. the probability that a sign will be observed by survey teams if it is fact present within in a given replicate). This is probably due to the fact that in the more rugged terrain which characterizes higher elevations both survey teams and ungulates tend to be more restricted in the routes that they can potentially use to traverse the landscape and thus tend to congregate along ridges or streams etc.

The modelling results suggested a significantly negative relationship between the occurrence of wild pig and domestic cattle, perhaps due to competition for similar resources. However, these analyses did not provide particularly strong evidence at the individual species level that occurrence is influenced by any of the other threat indicators measured. This is likely a paradoxical consequence of the paucity of data for the more hunting sensitive species (serow and sambar), as this lack of data greatly limits any attempt to model deterministic processes within a system. In contrast, when all ungulate observations are combined, which yields a far greater amount of raw data, the aggregate response variable is evidently strongly negatively influenced by the occurrence of domestic cattle and by the occurrence of snares, both of which conform to predictions. In addition, the responses described above with regard to elevation and proximity to settlements are also related to threats. Both are factors which relate to the accessibility of sites, which in turn influences the occurrence of threats. All of these results serve to corroborate the assertion that ungulate occurrence and abundance are to a large degree determined by the distribution and intensity of threats. Visual inspections of the species distribution and threat distribution maps also suggest a general pattern whereby less accessible areas have higher species values and lower threat values.

The presence of law enforcement patrols within an a site is expected to exert a deterrent effect but models including a measure of law enforcement effort (no. km patrolled per grid cell) did not yield a substantially better fit for any individual species. For all species combined these data suggested a positive relationship between the amount of patrol effort invested and the abundance of ungulates, which is contrary to predictions and initially seems counter-intuitive. However, the typically complex and non-linear nature relationship between enforcement effort, threats and species occurrence and abundance is well documented (Keane et al., 2011; O'Kelly, 2013). Law enforcement teams are by nature reactive; thus patrol effort will be focused on areas of high threat, presumably with threat levels consequently decreasing at some unknown rate. Wildlife abundances in these areas may be high or low, depending on how long threats have been present and/or how long they have been targeted by enforcement efforts. Lags of varying length may occur between patrols and any subsequent deterrent effect, and the duration of any such effect is unknown. The spatial scale at which any deterrence effect will operate at is also unknown and is likely to be dependent on a multitude of factors. In addition, law enforcement teams may be tasked with protecting areas of special importance for wildlife populations, i.e. where densities are high and threats are low. Thus, any attempt to quantify the impact of law enforcement activities is highly sensitive to the spatial and temporal scale of the analysis and multiple confounding factors are likely to obscure relationships of interest.

This is despite the fact that site specific spatial patterns seem to be apparent within the species distribution maps. For example; it seems probable that in PST high levels of patrol effort have been instrumental in reducing threats which in turn has contributed to the higher rates of occupancy and abundance values evident there. In contrast, the southeast sector of NKD has had little patrol effort, and threat occurrence values are high in that area, while species occurrence and abundance values are low. However, other spatial patterns are more difficult to interpret. In the northwest of NKD wildlife numbers appear to remain high, and threat occurrence values low, despite a lack of patrol effort.

5.3 Monitoring Indicators

The four ungulates selected as target species were chosen on the assumption that they are susceptible to the major threats affecting the majority of species within this landscape, namely hunting and, to a lesser extent, habitat loss and degradation. A corollary of this is that where and when management interventions are successful in mitigating these threats, ungulate populations should be able to recover relatively quickly. The most fundamental requirement for the purposes of monitoring is that such recoveries (or otherwise) should be identifiable and quantifiable using the method employed in this survey.

None of the species selected for this survey can be considered rare in comparison to other large mammal taxa (i.e. medium and large carnivores, primates etc). Such rare species may be of high conservation value but collecting sufficient data to reliable and precisely estimate and track population trends has been shown to be largely unfeasible in a Lao context (Hallam et al., 2015 *in press*). Yet neither could all target species included in this survey be considered common, and indeed for sambar and to a certain extent serow, observations were scarce, to the extent of precluding credible species specific estimates for sambar.

Given this scenario, a renewed focus on more abundance species such as pig and muntjac may seem appealing, but such an approach does involve inherent risk from a management perspective. Wild pig is a particularly good example of how indicator species may not always reflect parameters of interest within a managed system. As described above, pig populations may be relatively resilient to hunting, but this issue is compounded by the fact that any adverse effects of hunting pressure may be difficult to detect. Pigs are social animals and group size is highly variable but often large, and under high hunting pressure, many individuals might be removed before a reduction in group density or occupancy rates becomes apparent (because hunters are picking off single animals rather than groups).

Despite the above concerns, the index values derived from all four ungulate species combined does appear to exhibit a response to the presence of threats. Furthermore, these measures, particularly the occupancy rates, are estimated with a high level of precision, making it possible to reliable identify and track trends. Managers must be mindful of the above considerations for these kinds of biological monitoring indicators, and when interpreting apparent patterns they should also do so within the context of multiple sources of information, including law enforcement monitoring (i.e. SMART data) and anecdotal information. This is particularly important in such dynamic landscapes where species densities are very low. Nevertheless, the combined ungulate index value described here can represent an appropriate measure for assessing the impact of interventions in a robust and timely manner.

The inclusion of threat indicators as a specific target for data collection is an important feature of this survey. Monitoring of threats is a standard way of evaluating intermediate conservation outcome (Kapos et al., 2009; Salafsky and Margoluis, 1999) but it becomes an even more critical component of a monitoring regime where densities are at such low levels (i.e. sambar and serow) or when target species may persist despite relatively high levels of threat (i.e. wild pig and muntjac). Not only does the information obtained on threats data provide a means of assessing the impact of management activities where a response is not, or not yet, evident in wildlife populations, but it can also be incorporated directly into the population modelling process for key species, ultimately facilitating improved estimation.

5.4 Conservation Opportunities

With the possible exception of sambar, ungulate populations are persisting across all sites. Abundances are low, but it seems probable that distributions and abundances are primarily determined by the distribution and relative intensity of threats, particularly of hunting. Natural ungulate densities in Indochinese forests are, in fact, largely unknown, but given anecdotal historical accounts and the apparent suitability of habitat as assessed by experts (Duckworth and Hedges, 1998) it seems likely that there is considerable potential for recovery across all sites. Such recovery can best be facilitated by focusing on achieving a reduction in hunting pressure. This will require a continued strong commitment to law enforcement, with a particular focus on the apprehension and subsequent prosecution of poachers. Other core components of protected management, such as education, outreach and support for the development of alternatives should continue to be implemented in parallel. A comprehensive monitoring system is also essential to determine to what extent management interventions continue to be effective.

6 Recommendations

Provided below are recommendations for future iterations of the survey, potential further analysis of these data and suggested issues which may require further investigation by site-based teams.

6.1 Management Recommendations

- Two areas within NKD NPA appear to have particularly high abundance values for all ungulates. They are the north east sector which is separated from the rest of the area by the road, and the central sector immediately south of this (see Figure 7). Both of these sectors also appear to have low levels of threat and presumably this has contributed to these high abundance values. What is surprising, however, is that neither sector is extensively patrolled (or has been in the past year). In fact the northern sector has received no patrol effort at all. It could be that rugged terrain has impeded access by both patrol teams and local people, but in fact access to these sectors does not seem especially difficult. Further investigation into why these areas are essentially self-protecting may be warranted.

- In contrast to the above point, the southern sector of NKD NPA also receives little or no patrol effort but threat levels are extremely high. This is likely due to the fact this area lies very close a major market and is subject to a constant drain to supply this demand. Given limited resources it may be strategically more efficient *not* to allocate resources to this (and other) high threat areas but instead to focus protection efforts on areas of high abundance. However, this should be a specified management tactic, and it is not clear in this instance if this is the case.

- In PST ESCA patrol coverage is even and high across the entirety of the site. However, threat levels are evidently higher in the northern and eastern boundary sectors, where abundance values are also lowest. Reports from project staff have not given any indication that these north eastern sectors are especially problematic with respect to hunting or other threats, although this is apparent from these results. Further follow-up with site-based teams should be undertaken to investigate these areas.

- In NGS PF ungulate abundance seems to be highest in the areas close to the villages in the eastern sector of the area. This maybe partially due to the exceptionally rugged terrain of this site, including large areas of karst, which restricts access to western sectors. However, threats seem to be more evenly distributed across the entire site and this situation requires further investigation to gain a clearer interpretation.

- Also in NGS PF patrol coverage, with access facilitated by the reservoir, does not seem to penetrating into areas of either high wildlife abundance or high threats. The maps provided as a result of this survey will be useful is assisting the management team to address this issue.

6.2 Technical Recommendations

- Given the high level of investment required this kind of survey cannot be conducted annually. It is recommended that the survey is repeated at three to four year intervals, depending on funding availability and grant commitments. Extending the period between surveys may seem preferable given financial and other constraints but this does risk overlooking sudden species declines and extirpations, which are possible in such dynamic systems experiencing high levels of pressure.

- All of the models considered during this analysis were single-season models. Such models can be used to look at the level or patterns in occupancy at a single point in time. It is essentially a snapshot of the presence and absence of a species for a given time period, in this case the dry season of 2015. Following the implementation of future surveys, multi-season models can be investigated. These models allow for the explicit estimation of rates of colonization and extinction at grid cell level.

- Given the issues associated with using biological monitoring indicators, rather than landscape or umbrella species, or species of high conservation value, additional information on hunting should be collected. This will help to ensure that assumptions regarding population responses to threats and to management intervention designed to alleviate threats remain valid. Such information may come from independent hunting assessments or some analysis of LEM-type data (i.e. SMART data).

- Related to the point above, an interesting and potential useful extension of this analysis would be compare the results of this survey with the distribution and of intensity of threats as described by the SMART data collected by patrol teams. This would be done by calculating catch-per-unit-effort (CPUE) indices (i.e. no of snares encountered per kilometre patrolled) for a similar range of threats and mapping these indices. Note that while generating CPUE indices controls for variable effort it does not account for imperfect detection or non-random sampling (O'Kelly 2013).

- A further extension of this analysis would be to implement a Bayesian approach to fitting similar hierarchical occupancy and abundance models. This dataset is suitable for such an approach, which has the advantage of far greater flexibility in terms of model specification. For example; including team leader as a random effect, in order to address variability in detection probability between teams, would be trivial within a Bayesian analysis whereas it is not possible within this frequentist approach. Free software for Bayesian analysis is readily available (i.e. WinBUGS & JAGS) but their use requires a high level of technical and coding expertise.

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References

Bailey, L.L., MacKenzie, D.I., Nichols, J.D., 2014. Advances and applications of occupancy models. Methods Ecol. Evol. 5, 1269–1279. doi:10.1111/2041-210X.12100

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L., 2001. Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, USA.

Burnham, K.P., Anderson, D., 2002. Model Selection and Multi-Model Inference, 2nd ed. Springer.

Corlett, R.T., 2007. The Impact of Hunting on the Mammalian Fauna of Tropical Asian Forests. Biotropica 39, 292–303. doi:10.1111/j.1744-7429.2007.00271.x

Danielsen, F., Mendoza, M.M., Alviola, P., Balete, D.S., Enghoff, M., Poulsen, M.K., Jensen, A.E., 2003. Biodiversity Monitoring in Developing Countries: What Are We Trying to Achieve? Oryx 37, 407–409. doi:10.1017/S0030605303000735

Duckworth, J.W., Hedges, S., 1998. Tracking tigers: a review of the status of tiger, Asian elephant, gaur and banteng in Vietnam, Lao, Cambodia and Yunnan Province (China), with recommendations for future conservation action. WWF Indochina Programme, Hanoi.

Duckworth, J.W., Steinmetz, R., Pattanavibool, A., 2008. Serow *Capricornis milneedwardsii*

IUCN Red List Threated Species 2008 ET3814A1010185 http://www.iucnredlist.org/details/3814/0).

Fiske, I., Chandler, R., 2011. unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. J. Stat. Softw. 43, 1–23.

Harrison, R.D., Sreekar, R., Brodie, J.F., Brook, S.M., Luskin, M., O'Kelly, H.J., Rao, M., Scheffers, B., Velho, N., In press. Impacts of hunting on tropical forests in Southeast Asia. Conserv. Biol.

Kapos, V., Balmford, A., Aveling, R., Bubb, P., Carey, P., Entwistle, A., Hopkins, J., Mulliken, T., Safford, R., Stattersfield, A., Walpole, M., Manica, A., 2009. Outcomes, Not Implementation, Predict Conservation Success. Oryx 43, 336–342. doi:10.1017/S0030605309990275

Karanth, K.U., Nichols, J.D., Kumar, N.S., Link, W.A., Hines, J.E., 2004. Tigers and their prey: Predicting carnivore densities from prey abundance. Proc. Natl. Acad. Sci. U. S. A. 101, 4854–4858. doi:10.1073/pnas.0306210101

Karanth, K.U., Sunquist, M.E., 1992. Population Structure, Density and Biomass of Large Herbivores in the Tropical Forests of Nagarahole, India. J. Trop. Ecol. 8, 21–35. doi:10.1017/S0266467400006040

Kawanishi, K., Sunquist, M.E., 2004. Conservation status of tigers in a primary rainforest of Peninsular Malaysia. Biol. Conserv. 120, 329–344. doi:10.1016/j.biocon.2004.03.005

Keane, A., Jones, J.P.G., Milner-Gulland, E.J., 2011. Encounter data in resource management and ecology: pitfalls and possibilities. J. Appl. Ecol. 48, 1164–1173. doi:10.1111/j.1365-2664.2011.02034.x

Legg, C.J., Nagy, L., 2006. Why most conservation monitoring is, but need not be, a waste of time. J. Environ. Manage. 78, 194–199. doi:10.1016/j.jenvman.2005.04.016

Lindenmayer, D.B., Gibbons, P., Bourke, M., Burgman, M., Dickman, C.R., Ferrier, S., Fitzsimons, J., Freudenberger, D., Garnett, S.T., Groves, C., Hobbs, R.J., Kingsford, R.T., Krebs, C., Legge, S., Lowe, A.J., Mclean, R., Montambault, J., Possingham, H., Radford, J., Robinson, D., Smallbone, L., Thomas, D., Varcoe, T., Vardon, M., Wardle, G., Woinarski, J., Zerger, A., 2012. Improving biodiversity monitoring. Austral Ecol. 37, 285–294. doi:10.1111/j.1442-9993.2011.02314.x

MacKenzie, D.I., 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Academic Press.

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Andrew Royle, J., Langtimm, C.A., 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83, 2248–2255.

MacKenzie, D.I., Nichols, J.D., Sutton, N., Kawanishi, K., Bailey, L.L., 2005. Improving Inferences in Population Studies of Rare Species That Are Detected Imperfectly. Ecology 86, 1101–1113. doi:10.1890/04-1060

McComb, B., Zuckerberg, B., Vesely, D., Jordan, C., 2010. Monitoring Animal Populations and Their Habitats: A Practitioner's Guide, 1 edition. ed. CRC Press, Boca Raton, FL.

Nichols, J.D., Williams, B.K., 2006. Monitoring for conservation. Trends Ecol. Evol. 21, 668–673. DOI: 10.1016/j.tree.2006.08.007

O'Kelly, H.J., 2013. Monitoring Conservation Threats, Interventions and Impacts on Wildlife in a Cambodian Tropical Forest. (PhD Thesis). Imperial College London, Institute of Zoology ZSL, London.

O'Kelly, H.J., Evans, T.D., Stokes, E.J., Clements, T.J., Dara, A., Gately, M., Menghor, N., Pollard, E.H.B., Soriyun, M., Walston, J., 2012. Identifying Conservation Successes, Failures and Future Opportunities; Assessing Recovery Potential of Wild Ungulates and Tigers in Eastern Cambodia. PLoS ONE 7, e40482. doi:10.1371/journal.pone.0040482

Pollock, K.H., Nichols, J.D., Simons, T.R., Farnsworth, G.L., Bailey, L.L., Sauer, J.R., 2002. Large scale wildlife monitoring studies: statistical methods for design and analysis. Environmetrics 13, 105–119. doi:10.1002/env.514

Salafsky, N., Margoluis, R., 1999. Threat Reduction Assessment: a Practical and Cost-Effective Approach to Evaluating Conservation and Development Projects. Conserv. Biol. 13, 830–841. doi:10.1046/j.1523-1739.1999.98183.x

Srikosamatara, S., 1993. Density and Biomass of Large Herbivores and Other Mammals in a Dry Tropical Forest, Western Thailand. J. Trop. Ecol. 9, 33–43.

Thompson, W.L., White, G.C., Gowan, C., 1998. Monitoring Vertebrate Populations, 1st ed. Academic Press, U.S.A.

Timmins, R., Kawanishi, K., Giman, B., Lynam, A.J., Chan, B., Steinmetz, R., Sagar Baral, H., Kumar, N.S., 2015. Sambar *Rusa unicolor* IUCN Red List Threaten Species 2015 ET41790A85628124. http://www.iucnredlist.org/details/41790/0.

Vongkhamheng, C., Johnson, A., Sunquist, M.E., 2013. A baseline survey of ungulate abundance and distribution in northern Lao: implications for conservation. Oryx 47, 544–552.

Williams, B.K., Nichols, J.D., Conroy, M.J., 2002. Analysis and Management of Animal Populations, 1st ed. Academic Press, San Diego, California, USA.

Yoccoz, N.G., Nichols, J.D., Boulinier, T., 2001. Monitoring of biological diversity in space and time. Trends Ecol. Evol. 16, 446–453. DOI: 10.1016/S0169-5347(01)02205-4