# INFO ENTRY - QUESTION INFO

## Topic Info

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| **info\_id** | objective |
| **question** | NULL |

One of the first choices you will make when designing your study, is what it is that you plan to measure, or your “state variable”. Since can easily be confused with objective

# DEFINTIONS

## Species inventory

## Species diversity & richness

\*\*{{ term\_mod\_divers\_rich }}\*\*: {{ term\_def\_mod\_divers\_rich }}:

Note that there are multiple “levels” to Species diversity & richness (**α-richness [alpha], γ-richness (gamma), and** **β-diversity (beta)**); refer to (Models - Species diversity & richness)(#i\_mod\_divers\_rich) for more details.

**Species richness:** the number of species found in the community/area measured (Pyron, 2010)

Species diversity:

**Alpha richness (α-richness):** species richness at the level of an individual camera location

**Gamma richness (γ-richness):** species richness across a whole study area

**Betadiversity (β-diversity):** the differences between the communities or, more formally, the variance among the communities {{ ref\_intext\_wearn\_gloverkapfer\_2017 }}

“It’s important to note that the scale over which species richness is calculated can affect the conclusions drawn, and may make it difficult to compare estimates from different studies. Some camera trap studies calculate species richness at the level of an individual camera location – often called α-richness (alpha richness) – whilst other studies calculate species richness across a whole study area – often called γ-richness (gamma richness). The scale used is important because of the species-area relationship: species richness accumulates as the area covered increases, but the rate of this increase may vary in different study sites, meaning that conclusions about which study site is the most diverse may change with scale.” {{ ref\_intext\_wearn\_gloverkapfer\_2017 }}

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6-2 Community variance or β-diversity

When considering two (or more) communities, it is possible to calculate a state variable which reflects the differences between the communities or, more formally, the variance among the communities. We sometimes call this community variance “β-diversity” (betadiversity). This is useful, for example, for assessing the degree to which communities subject to different management differ (e.g. comparing an old-growth site, a selectively-logged site and a plantation forest site). This is sometimes called “across-site” β-diversity, because it is being used to assess community variance across heterogeneous habitat types.

β-diversity measures can also be used to assess community variance within single habitat types, at a smaller scale. This is sometimes called “within-site” β-diversity (although the distinction from across-site β-diversity may not always be clear-cut). This can be important because changes in community variance within a study site may reflect changes in the fundamental processes which generate biodiversity at local scales (such as habitat heterogeneity and the connectivity of populations).

β-diversity should also play an important role in spatial conservation planning, for example in designing networks of reserves. All else being equal, if β-diversity is high, it will be important to establish a network of reserves so that all species in the landscape are covered. On the other hand, if β-diversity is low and communities are similar across space, then a single large reserve may be the best option.

Communities can also be compared across time, rather than across space, giving rise to temporal β-diversity. This can be used to track how much, and how quickly, communities are changing at a single site over time.

Camera trap studies typically sample a large number of locations, making them highly suitable for quantifying β-diversity, but this has rarely been done (but see: Wearn et al. 2016). At least in part, this is probably because the importance of β-diversity is poorly appreciated amongst wildlife biologists and conservationists. In addition, there are many different ways β-diversity can be calculated, each with their own strengths and weaknesses, with no single best measure. This can be confusing and lead to “analysis paralysis”. In common with species richness, β-diversity is also dependent on spatial scale (Olivier & Aarde 2014). For example, some habitats such as logged forests may show high β-diversity (rapid community turnover) at fine spatial scales, but low β-diversity (homogenous communities) at coarse spatial scales (Wearn et al. 2016). Finally, interpreting and communicating measures of β-diversity can be hard, because they are often in meaningless units, or because they do not lend themselves directly to comparisons across different studies.

## Occupancy

the probability a site is occupied by the species {{McKenzie et al., 2002}}. Occupancy is also highly suitable for evaluating broad-scale patterns of species distribution {{Wearn & Glover-Kapfer, 2017}}.

## Abundance (Relative vs. Absolute)

Abundance is a deceptively difficult state variable to measure. This applies to camera traps just as it does to other methods. A huge benefit of camera traps is that they are continuously running, and it is possible to count every individual animal that walks past a camera, unlike say a single-catch live trap which effectively stops recording once it has caught one animal. However, it is usually difficult to tell individual animals apart in camera trap images, so that over the course of a week’s sampling you often don’t know if you have 10 captures of one individual or one capture for each of 10 different individuals. As for species richness, abundance measures are also affected by imperfect detection, so that even if you could tell individuals apart, you will likely have missed some individuals in the population. This is the case especially using camera traps, which typically each “see” only a tiny 100 m2 portion of the ground, and even less in dense vegetation or if the ground surface is not flat. Even if you were to add up the area covered by all of your cameras, and all of the time that they have spent “watching” for individuals, you are unlikely to have achieved a full census of the population.

There are two broad approaches to these fundamental problems: 1) control the sampling methods as much as possible and use an **index of true abundance**, or 2) explicitly try to model the process by which animals are detected and obtain an estimate of **absolute abundance.** We focus here on abundance (the number of individuals in a population), and then deal with density (the number of individuals per unit area).

### Relative abundance

is an an indirect measure of abundance

Relative abundance can be evaluated via “indices”

When observational data is converted to a detection rate (i.e., the frequency [count] of independent detections of a species within a distinct time period). An index can be a count of animals or any sign that is expected to vary with population size (Caughley, 1977; O’Brien, 2010).

**Intensity of use**

“the expected number of use events of a specific resource unit during a unit of time” (i.e., “how frequently a particular resource unit is used”) (Keim et al., 2019).

“Intensity of use differs from probability of occupancy, selection or use, which can remain constant even when the intensity of use varies” (Keim, DeWitt, & Lele, 2011; Lele et al., 2013).

**Probability of use**

“the probability of at least one, use event of that resource unit during a unit of time” (i.e., “would a particular resource unit be used at least once) (Keim et al., 2019).

The simplest way of analysing the data is to use the frequency of animal detections, or trapping rate (typically, detections per 100 nights of camera trap sampling), as an indirect measure of abundance (see Box 6-2 for how detections are counted in practice). This is often referred to as relative abundance – to distinguish it from actual absolute abundance – and the resulting measure is often called a relative abundance index (RAI). Clearly, trapping rates are going to be influenced by much more than just the abundance of animals. For this reason, they have been highly controversial (e.g. Anderson 2001; Sollmann et al. 2013c). For example, trapping rates will be affected by how active animals are (animals which are active for longer or cover more ground will trigger the cameras more) and how large they are (animals which are larger are more likely to be detected by the passive infrared sensors on most camera traps). However, attempts can be made to standardise at least some of the factors that affect trapping rates by very carefully designing the study (see Chapter 7-6). It is also possible to estimate the size of the detection zone of the camera traps in different habitats or for different species and apply corrections to the indices (Rowcliffe et al. 2011).

A simple way of estimating the effective detection distance (i.e. the detection zone’s radius) is to place markers in the field of view at known distances, and then record the approximate distance at which animals are detected (Caravaggi et al. 2016; Hofmeester et al. 2017; Fig. 6-1). The effective detection distance can then be estimated using distance sampling methods (Hofmeester et al. 2017). A short cut to controlling for variation in detection distances is to only count animal detections within a short distance that is unobstructed and well sampled across all cameras and all species (e.g. 3 m, indicated by a marker placed in the field of view). However, this will necessarily involve discarding a portion of the dataset.

Despite the controversy that RAIs invoke, their judicial use can still offer meaningful insights into wildlife populations. In addition, where RAIs have been compared to robust density estimates, the correlations across space (Rovero & Marshall 2009), across studies (Carbone et al. 2001), and even across species (O’Brien et al. 2003; Rowcliffe et al. 2008; Kinnaird & O’Brien 2012) have usually been positive and apparently linear.

## Population size (Absolute abundance): the number of individuals in a population (Wearn & Glover-Kapfer, 2017).

6-3-2 Capture-recapture, mark-resight and the Royle-Nichols model

The second broad approach to estimating abundance from camera trap data is to attempt to describe, using a model, the ecological and methodological processes which gave rise to the data. By doing this, it is possible to obtain an estimate of absolute abundance, i.e. the number of individuals in a population. This could be the number of individuals in a fenced reserve or on an island. Note that in a continuous habitat, what a “population” is can be difficult to define, and density is often a more useful target for monitoring.

## Density : The number of individuals per unit area (Wearn & Glover-Kapfer, 2017).

## Vital rates

(e.g., survival probabilities and recruitment rates)

## Behaviour

behaviour focused objectives vary greatly; they may be qualitative or quantitative (Wearn & Glover-Kapfer, 2017) (e.g., diel activity patterns, mating, boldness, predation, foraging, activity patterns, vigilance, parental care [Caravaggi et al. 2017; Wearn & Glover-Kapfer, 2017]).

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| Species inventory |  |
| Species diversity & richness |  |
| Species diversity |  |
| Species richness |  |
| Species diversity & richness - α-richness (alpha richness) |  |
| Species diversity & richness - γ-richness (gamma richness) |  |
| Species diversity & richness - β-diversity (betadiversity) |  |
| Occupancy |  |
| Relative abundance |  |
| Relative abundance - Intensity of use |  |
| Relative abundance - Probability of use |  |
| Population size |  |
| Absolute abundance |  |
| Vital rates | (e.g., survival probabilities and recruitment rates) |
| Density |  |
| Behaviour |  |

## Overview

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## Advanced

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## Figures

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## Analytical tools & resources

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## References / Glossary

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## Notes