

## Chapter 2

# Global Terrestrial Ecosystem Observations: Why, Where, What and How?

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**Abstract** This chapter covers the questions of ecosystem definition and the organisation of a monitoring system. It treats where and how ecosystems should be measured and the integration between in situ and RS observations. Ecosystems are characterised by composition, function and structure. The ecosystem level is an essential link in biodiversity surveillance and monitoring between species and populations on the one hand and land use and landscapes on the other. Ecosystem monitoring requires a clear conceptual model that incorporates key factors influencing ecosystem dynamics to base the variables on that have to be monitored as well as data collection methods and statistics. Choices have to be made on the scale at which monitoring should be carried out and eco-regionalisation or ecological stratification are approaches for identification of the units to be sampled. This can be done on expert judgement but nowadays also on stratifications derived from multivariate statistical clustering. Data should also be included from individual research sites over the entire world and from organically grown networks covering many countries. An important added value in the available monitoring

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technologies is the integration of in situ and RS observations, as various RS technologies are coming into reach of ecosystem research. For global applications this development is essential. We can employ an array of instruments to monitor ecosystem characteristics, from fixed sensors and in situ measurements to drones, planes and satellite sensors. They allow to measure biogeochemical components that determine much of the chemistry of the environment and the geochemical regulation of ecosystems. Important global databases on sensor data are being developed and frequent high resolution RS scenes are becoming available. RS observations can complement field observations as they deliver a synoptic view and the opportunity to provide consistent information in time and space especially for widely distributed habitats. RS has a high potential for developing distribution maps, change detection and habitat quality and composition change at various scales. Hyperspectral sensors have greatly enhanced the possibilities of distinguishing related habitat types at very fine scales. The end-users can use such maps for estimating range and area of habitats, but they could also serve to define and update the sampling frame (the statistical ‘population’) of habitats for which field sample surveys are in place. Present technologies and data availability allow us to measure fragmentation through several metrics that can be calculated from RS data. In situ data have been collected in several countries over a longer term and these are fit for statistical analysis, producing statistics on species composition change, habitat richness and habitat structure. It is now possible to relate protocols for RS and in situ observations based on plant life forms, translate them and provide direct links between in situ and RS data.

**Keywords** Ecosystem monitoring • Habitat • Hyperspectral sensor • In situ observation • Plant life form • Stratification • Sensor networks

## 2.1 Introduction

In the last decades it has been emphasised that we still lack empirical baseline data on local patterns of biodiversity and their dynamics and interactions within communities and habitats (Scholes et al. 2008). The lack of empirical biodiversity observation data is obvious at various levels of complexity; even basic inventories of current local-to-global biodiversity are missing. There are several reasons for this. Firstly, global cooperation in biodiversity research and monitoring is a relatively recent phenomenon. We lack standards, we do not yet share protocols, we do not consider strategic sampling and there is limited exchange of data at and between spatial scales. Noss (1990) flagged this problem and developed a general concept for a hierarchical approach to monitoring biodiversity. Ecosystem monitoring is needed to track the impacts of various drivers such as land use change and climate change.

In this chapter we deal with dryland terrestrial ecosystems (marine and freshwater species and ecosystems are dealt with in Chaps. 7 and 8, respectively).

Several long term ecosystem monitoring networks, based on coordinated long-term observation systems, do exist. Examples include the networks of the International Long-Term Ecological Research Network (ILTER/LTER; global, national scale), the Important Bird and Biodiversity Areas program (IBA; global scale), the Biodiversity Monitoring Transect Analysis in Africa project (BIOTA; Africa, São Paulo State), the Global Observation Research Initiative in Alpine Environments program (GLORIA; mountain summits at a global scale), the South African Environmental Observation Network (SAEON; South Africa), the Federal System of Protected Areas (SiFAP; Argentina), the Terrestrial Ecosystem Research Network (TERN; Australia), the National Ecological Observatory Network (NEON; USA), and the Amazon Forest Inventory Network (RainFor; Amazon).

This Chapter Covers Four Main Issues and Comprises Four Sections:

- what is an ecosystem?
- where to measure ecosystems,
- what to measure and how to measure it, and
- how to link the various approaches and protocols.

All four issues require choices by decision-makers concerning effort, budget, human resources and infrastructural capacities.

## 2.2 Ecosystems and Ecosystem Variables

Ecosystems are universally understood as systems of biotic communities interacting with themselves and with their abiotic environment. Ecosystems can be conceptualised as the integration of living and non-living components in nature. They are characterised by their composition, function and structure which depends on the local environment, as well as management approaches. Each of these three dimensions should be included in ecosystem monitoring.

In biodiversity surveillance and monitoring, ecosystems are an essential link between species and populations on the one side, and land use and landscapes on the other (e.g., Noss 1990; also see Fig. 1.1 in Chap. 1). What could be measured in ecosystems potentially touches on all the major dimensions of biodiversity. Therefore strategic choices have to be made about what should be measured, and how and where to measure it.

Ecosystems in the most general sense are conceptual rather than physical entities, and are therefore dimensionless. Their spatial or structural aspects do have physical manifestations, with units, and can be defined as ecotopes or habitats. Definitions of the term ‘habitat’ range from how species are associated with landscape-scale units to very detailed descriptions of the physical environment used by species (Hall et al. 1997). They also include aspects such as snow cover, openness and patchiness. Bunce et al. (2008) gave a practical definition of habitats and rules for assignment of a given patch to a habitat class. They define habitat as *‘an element of the land surface that can be consistently defined spatially in the field in order to define the principal*

*environments in which organisms live*'. Functional aspects of ecosystems can be defined as the cycling of matter and energy expressed in biomass, seasonal changes, succession and soil development, growth, energy storage and regulation processes. Compositional aspects of ecosystems are species richness, diversity of species and guilds, and presence of certain species assemblages.

In many cases ecosystems and habitats are, in practice, defined based on their vegetation compositional and/or structural aspects. Classical phytosociology was designed for description, rather than long-term monitoring and change detection, but individual plots that have been studied in the past can be resampled, if the sites are re-locatable. Vegetation structure and biomass are more important for animal populations than vegetation composition and some widely recognised habitats may not be directly linked to vegetation composition. The TERN project ([www.tern.au](http://www.tern.au)) stipulates that a monitoring design needs to pay careful attention to:

- the question(s) of interest;
- statistical principles;
- a conceptual model that incorporates the key factors influencing ecosystem dynamics;
- the type of entities that need to be monitored;
- the data collection methods that will be effective; and
- the scale of the required monitoring program.

It is important to realise that errors are inevitable and that in some cases absence of a feature (a dry lake with no water) or taxon (no birds in a forest) is as important as its presence. Measuring a non-stable variable that may be associated with a particular error to boot, can lead to a poor level of understanding. In other words, too many constraints in a monitoring scheme may reduce the likelihood of a monitoring system being successful. Therefore an appropriate and sound statistical design that, for instance, can deal with variability and the presence of null records (zeros) is essential in the set-up of long-term monitoring schemes.

Because we are interested in detecting trends, long-term quantitative approaches in measurement are important. There are many different variables that could be measured, so choices have to be made. Land cover forms a valuable basis for practical applications like forest and rangeland monitoring, but also for monitoring climate change, biodiversity and desertification (Jansen and Di Gregorio 2002). Climate and agricultural variables are measured under the umbrella of the World Meteorological Organisation (WMO) and the Food and Agriculture Organization of the United Nations (FAO) respectively. The key variables to be measured for biodiversity are variables related to ecosystem status and trends.

After the Nagoya Conference of the Parties of the Convention on Biological Diversity (CBD), the Group on Earth Observations Biodiversity Observation Network (GEO BON) organised a series of workshops to assess the possibility of collecting data relevant to reporting on progress in reaching the targets of the convention. In the process GEO BON developed the concept of Essential Biodiversity Variables (EBVs; Table 2.1; Pereira et al. 2013).

**Table 2.1** Some candidate ecosystem related Essential Biodiversity Variables (EBVs)

Class	Candidate EBV	Example parameters	Definition
Ecosystem process/function	Photosynthetic activity	Forest tree diameter, biomass production, sphagnum growth, ocean plankton	Fixation of carbon and production of O <sub>2</sub> by the biosphere
	Aerobic respiration	Autotrophic and heterotrophic respiration from soil and vegetation	Uptake of oxygen and production of CO <sub>2</sub> by the biosphere
	Phosphorus tolerance/limitation		Phosphorus load in an ecosystem in relation to a reference condition
	Secondary production	Fish population, krill density, wildebeest population, migratory bird population	Sum of production by herbivores and higher trophic levels
	Nutrient retention	Nitrogen retention	Capacity of ecosystem to store, fix or retain a given nutrient or chemical
	Disturbance regime	Wind-blow, hurricanes, el Niño, forest fires, flooding	Type, timing and intensity frequency of external influences on ecosystems
Ecosystem structure (mean and seasonality)	Cover, amount per volume, height clumping	Fractional cover, vegetation height	Organisation of habitats in a three dimensional space
	Ecosystem extent and distribution	Land cover	Total occupied area by type
	Ecosystem connectivity and fragmentation	Landscape heterogeneity	Connectivity is defined by patch size, shape, pattern and the ability to move between patches
Ecosystem composition (in space and time)	Community or functional composition	Species or functional profile based on relative abundance	Species or functional profile based on relative abundance
	Phylogenetic diversity (PD) of community		Sequence data based on phylogeny is used to assess genetic sequence based diversity in communities
Ecosystem level responses	Degree of protection		Level of ecosystem extent protected (through legal status, actual management and ownership)
Ancillary attributes of ecosystems	Richness and endemism		Sum of known species within an area (or ecosystem) and of species endemic to that area (or ecosystem)
	Uniqueness		Percentage of the global extent of an ecosystem found in a certain area

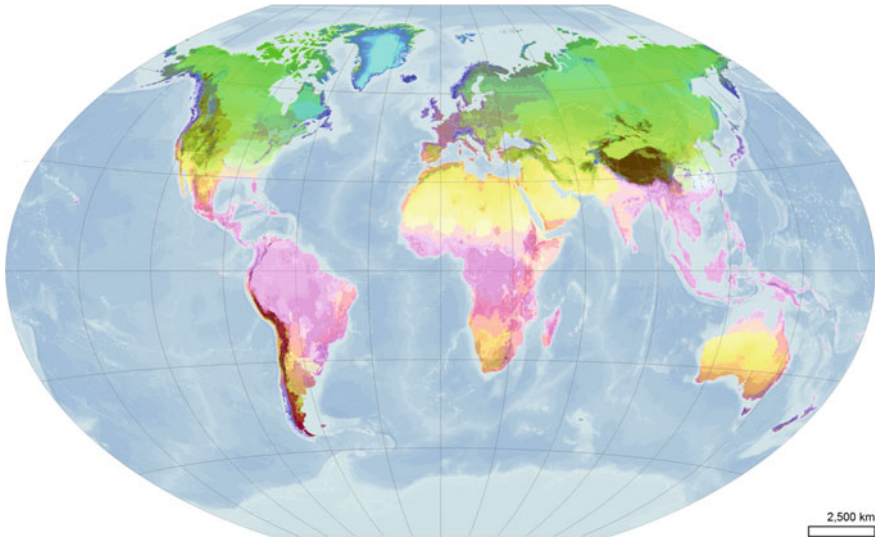
## 2.3 Where to Measure Ecosystem Variables

The question of where to measure ecosystems and ecosystem variables for an analysis at a particular scale calls for a ‘sampling frame’ that is strategically located across the globe, continent, country or region. The use of remotely-sensed land cover maps provides the first part of the picture of habitat change. It will therefore be an important tool for reporting change.

In addition to the overview of structural ecosystem change provided by repeated habitat maps there is a need for statistics on change and a need for monitoring of ecosystem processes. Here the question of where to measure becomes critical. For many purposes, such as consistent input to climate impact models, or reporting towards the Aichi targets, standardised frameworks and methods are required among different studies or countries to enable integration of data and reporting. The development and adoption of harmonised methods is a complex and difficult process, because ecological data collection tends to be coordinated at the regional or national level, following country specific methods, classifications and priorities. It is made more difficult by the long-term nature of the data: it may not be possible to harmonise data from old studies, and those responsible for the collection and curation of long-term records are typically reluctant to change their methods in substantive ways.

Ecosystems can be as extensive as the entire arctic tundra, or as small as a particle of soil. They are thus understood to exist at multiple scales. This means that choices have to be made on the scale at which monitoring should be carried out. Mapping ecologically homogenous regions across the planet to select monitoring sites has been accomplished through a process of eco-regionalisation as in the WWF global ecoregions map. However, this and most other approaches rely heavily on expert judgement for interpreting class divisions. This makes it difficult to ensure reliability across the world and limits their use in scientific analysis. The Global Environmental Stratification (GEnS) is the first high-resolution global bio-climate stratification derived from multivariate statistical clustering (Fig. 2.1). The GEnS also provides sufficient detail to support the design of regional monitoring programmes that can be nested within the global network.

A cost-efficient and data-effective selection of sites for data collection should be based on a stratified random selection procedure for the whole land surface of the target area. The GEnS (Fig. 2.1) is a way to provide a common global framework for positioning fixed monitoring stations, the development of LTER sites as well as for stratified random sampling and global statistics (Metzger et al. 2013a). The GEnS consists of 125 strata, which have been aggregated into 18 global environmental zones. The stratification has a 30 Arcsec resolution (equivalent to 0.86 km<sup>2</sup> at the equator). One of the recent applications of the GEnS is the ecological monitoring project in the Kailash Sacred Landscape (KSLCI). This is the first

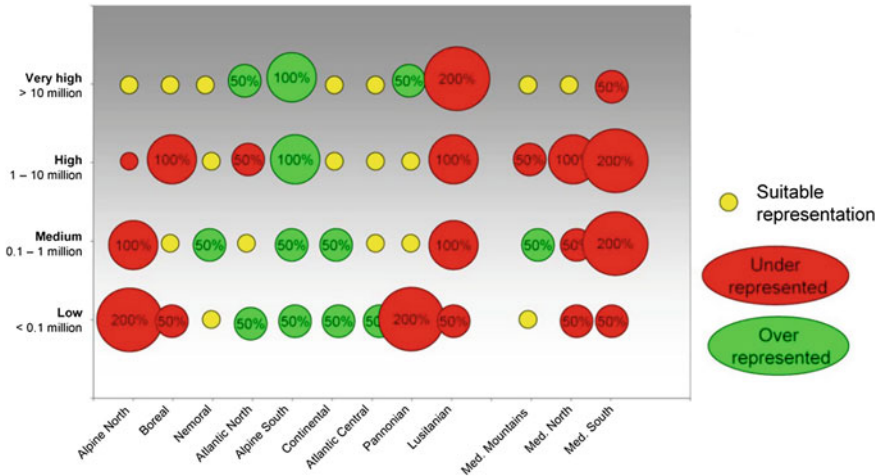


**Fig. 2.1** Global environmental zones map derived from temperature, precipitation, and seasonality data and with a grid of 30 Arcsec squares. The stratification exists of 125 strata in 18 global zones. *Source* Metzger et al. (2013a)

cooperation of its kind among China, India, and Nepal seeking to conserve the area through application of transboundary ecosystem management and enhanced regional cooperation (Metzger et al. 2013b). A comparable ecoregion based approach has been used in the USA to identify the NEON monitoring sites. The outcome of the geographical analysis resulted in twenty domains in which the observatories have been placed.

Data are collected at individual research sites or by national monitoring systems, all over the world. This process is currently not globally coordinated. The Long Term Ecological Research sites network (LTER) in Europe is an example of an organically grown network that covers many countries. There are at present approximately 1000 facilities with LTER activities, ranging in extent from less than 10 ha to several thousand hectares. They differ in monitoring objectives, methods of measurements, and spatial extent. However, as Metzger et al. (2010) showed, their distribution is not even (Fig. 2.2).

One may of course question whether one site per region can adequately address the eco-climatic variability in a large, diverse areas. In the NEON design this problem has been tackled by including both permanent core sites and relocatable auxiliary sites that should allow for covering the variation within a region. Remote sensing observations can allow generalisation of point samples over larger areas.

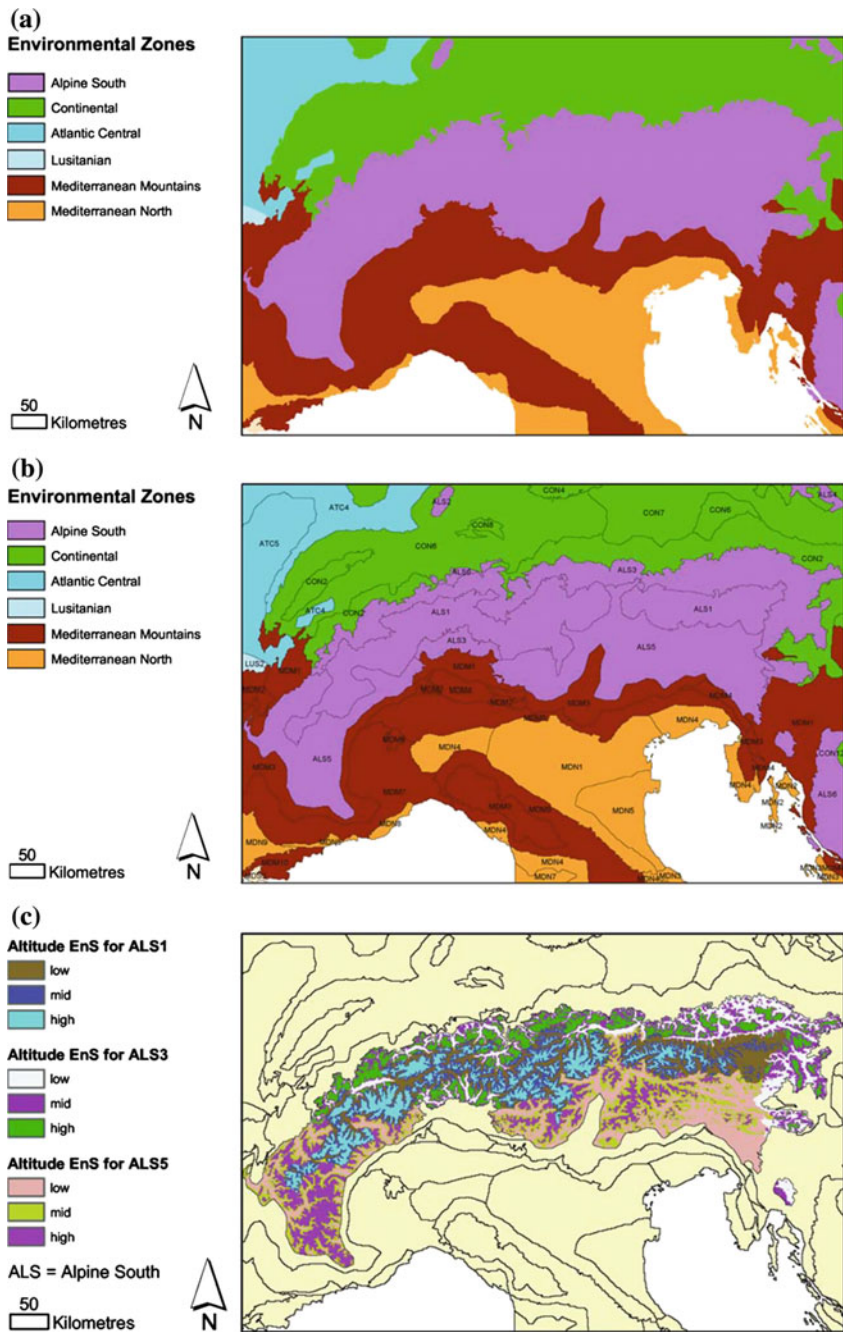


**Fig. 2.2** Representation of LTER facilities per socio-ecological region based on the Environmental Stratification of Europe. The strata in the X-axis are European Environmental Zones; the Y-axis indicates population density. *Source* Metzger et al. (2010)

The BIOTA observatories in Africa (Morocco, West Africa and South Africa) are situated on transects and each consists of a series of 1 km<sup>2</sup> squares where species and ecosystem variables are measured regularly (Jürgens et al. 2011). They also provide ground-truthing for remote-sensing observations. In this example, several ‘auxiliary observatories’ have also been established at a variety of scales, for process and pattern observations.

In global and continental stratifications climate plays a dominant role. This changes when stratifications are made at national and regional scales, especially in smaller countries and mountainous areas. Then the stratification should be broken down in a hierarchical flexible structure. In Fig. 2.3 such an approach is shown for the Alpine region in Europe. In Fig. 2.3a the Alpine region is shown in an aggregated way, and consists of large climate zones. This level is appropriate for reporting at the European level. Figure 2.3b shows the Alpine zone at the more detailed level of environmental strata (ALS1, ALS3 and ALS5) based on mainly climate variables. At this level, summits, valley sides and valley floors are still included in the same stratum, because of the smoothing effect of the climate data. The ecosystems and taxa in these different topographic locations will be very different. Therefore a subdivision based on altitude is made (Fig. 2.3c). This demonstrates the full complexity of the Alpine zone and will enable any sample of 1 km<sup>2</sup> plots to be dispersed efficiently through the landscape, i.e., on valley floors, valley sides and summits. At an even lower level, not only geomorphology, but also other information such as soil types and hydrology can be used for further refinements.





**Fig. 2.3** **a** Alpine regions according to division in environmental zones; **b** Alpine zone subdivided in environmental strata (ALS1, ALS3 and ALS5) within Alpine zone; **c** Alpine zone with environmental strata subdivided according to altitudinal bands. *Source* Jongman et al. (2006)

## 2.4 How to Measure Ecosystem Variables

There are generally three ways to measure ecosystem variables.

1. Most of the functional processes can be measured as fluxes, using in situ sensors.
2. Precise monitoring of composition, abundance, extent and change is commonly done by in situ monitoring through habitat surveillance combined with vegetation plots.
3. Structural change is monitored using in situ habitat surveillance in combination with remote sensing from space or aircraft.

There are advantages and differences between the methodologies and one solution does not satisfy all data questions. Remote sensing technologies are increasingly becoming integrated with in situ measurements as various new technologies become available for ecosystem research. For global applications this development is essential. Nowadays we can employ an array of instruments to monitor ecosystem characteristics, from fixed sensors and in situ measurements, to drones, planes and satellite sensors (Fig. 2.4).

### 2.4.1 Sensor Networks

Biogeochemical components determine much of the chemistry of the environment (air, water, and soil) and the geochemical regulation of ecosystems. Key measurements, among others include the greenhouse gases  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{NO}_x$ , which



**Fig. 2.4** An overview of the available array of sensors to measure ecosystem variables and metrics, varying from in situ sensors and surveillance to drones, airplanes and satellites

determine the climate change process and are important drivers of change in biodiversity. These and other chemicals such as  $\text{NH}_4$  also can cause acidification and eutrophication and in this way lead to ecosystem degradation, involving a sustained loss of ecosystem services and/or biodiversity. The water, carbon and nitrogen cycles have a direct influence on ecosystems globally and are measured using sensor networks in many countries in the world. Long-term, patch-scale measurements using eddy covariance (EC) are, for example, employed to estimate ecosystem carbon budgets. This is mainly done in research sites or dedicated monitoring sites. A global database of soil respiration data has been developed by the US Oak Ridge National Laboratory (<http://daac.ornl.gov>). It can be used as a reference database, because the number of sites is small, but it covers the globally important terrestrial ecosystems.

The extent to which pollutants are detrimental to ecosystem function and biodiversity is not always known, but clear effects have been reported for nitrogen, phosphorus, sulphur, pesticides, herbicides, aerosols and ozone. For an indication of excess pollutant exposure, it is important to know the difference between natural versus anthropogenic exposure levels. For this purpose emission, dispersion and deposition model calculations are generally used. Measurements of pollutants are made in many countries, but mostly at irregular intervals and patchily over space. Global coordination and harmonisation are lacking, but there are attempts to improve this, for instance in the way nitrate is measured in networks in Europe (EMEP), North-America (NADP), Canada (CapMon), and East Asia (EANET).

### 2.4.2 *In Situ Mapping*

Common approaches for in situ monitoring of ecosystem extent require definitions that are harmonised nationally, continentally and globally, which is not the situation at present. Forest definitions differ between international organisations such as FAO, CBD and UNFCCC and between European countries.

Surveillance involves recording of features at a specific location at one moment, i.e., taking stock. Monitoring involves repeated observation, to create a time series which enables the detection of change. This requires that the location of monitoring is known, and preferably kept constant over time. Moreover, in most cases the field assessment of biodiversity or habitats is based on samples. Sampling procedures must not be compromised by spatial heterogeneity or complexity. As sampling effort (i.e., the time taken to record information) is usually fixed, a choice has to be made between recording basic information in many sample units, or more detailed information in fewer units; similarly there is a trade-off between many small and few large units (Bunce et al. 2008). This has consequences for the statistical inference which can be made using the data. Often the optimal solution is neither one nor the other, nor an intermediate state, but a clever combination which has many simple sites for extrapolation purposes and a few comprehensively monitored sites to understand the details and processes.

For recognising trends and sudden changes in ecosystem composition and diversity it is important to produce statistics based on direct measurements. These can be used to derive indicators such as pattern and changes in species richness, patchiness and linear features. This has been done in the Great Britain Countryside Survey since 1978, producing statistics on species composition change, habitat richness and habitat structure to support policy ([www.countrysidesurvey.org.uk/](http://www.countrysidesurvey.org.uk/)). The configuration and fragmentation of structural biodiversity, species composition, age of systems and their components as well as biomass, ecological relations and extinction rates are important aspects related to ecosystem health and integrity.

For statistically-robust trend detection it is essential to return periodically to the same sites to record changes. National and regional in situ networks exist for monitoring ecosystems and biodiversity change. They employ various size units from 16 km<sup>2</sup> down to 0.25 km<sup>2</sup>. Some, such as the META project in Hungary, use hexagonal units of 35 ha, because a hexagon has six neighbouring cells with all more or less the same distance from the centre (Molnár et al. 2007). The most common emerging scale for the field recording of habitats is 1 km<sup>2</sup>, making a compromise between detail and generality.

In the EU-FP7 EBONE project a habitat and vegetation recording procedure was elaborated and made generally available ([www.wageningenur.nl/ebone](http://www.wageningenur.nl/ebone)). It includes a manual and a database with a digital field form that helps to support consistent mapping. The protocols have adopted plant life forms as the basis of a system of General Habitat Categories (GHCs). The GHC system includes some classes such as mud flats and scree slopes which do not have vegetation, in order to cover the terrestrial world from forests and grasslands to deserts. At a continental level, ecosystems can best be defined in terms of the physiognomy and life forms of the dominant species, because individual species are too limited to encompass widely dispersed geographical locations. Moreover, life forms can provide direct links between in situ and remotely-sensed data and dynamic global vegetation models. GHCs have been tested successfully throughout Europe, Israel, South Africa and Western Australia. The GHC framework also made it possible to harmonise different national habitat mapping systems so that they could be used to produce consistent indicator information across borders. It is therefore a good candidate to be tested globally.

### **2.4.3 Remote Sensing**

Traditionally, ecologists map biodiversity and ecosystems based on in situ observations, perhaps generalised using aerial photography. However, existing Remote Sensing (RS) tools can be used to measure and map a number of ecosystem variables and metrics directly, much more effectively than can be done using field measurements. RS is recognised as a powerful tool to acquire synoptic data on habitats, but to date, its use for operational monitoring and reporting of biodiversity is still limited. One reason for this appears to be the knowledge gap between the

agencies and individuals responsible for biodiversity monitoring and the remote sensing community. To overcome this gap requires mutual awareness, willingness to collaborate and technology transfer.

RS observations can complement field observations as they deliver a synoptic view and offer the opportunity to provide consistent information in time and space (Vanden Borre et al. 2011). It must be determined in each case what variable can be measured best by using RS, alone or in a hybrid scheme with an optimally-distributed set of in situ measurements. Recognition of habitat types on images is easier for widely distributed habitats than for rare habitat types. In general, rare ecosystems have to be specially targeted and small habitat elements (smaller than the minimum resolution of space-based sensors, which is in the region of 1–5 m for non-military instruments, and down to 0.3 m using airborne sensors) can only be monitored by in situ observations. Habitat distribution maps, change detection and even habitat quality and composition change at various scales can be cost-effectively monitored with these types of sensors (Turner et al. 2003). Although these techniques are promising, they still fall short in several aspects (Mücher et al. 2013): (i) airborne hyperspectral data or airborne Lidar are suitable, but coverage is still limited; (ii) existing methods have not fully addressed the issue of habitat structure and functioning, which is a key factor for assessing habitat quality; and (iii) most existing remote sensing methodologies have not been tested rigorously for operational purposes.

Monitoring of habitat quality information in enough detail remains challenging as this requires sensors and methods which can deal with complex transitional gradients in natural vegetation. Hyperspectral sensors offer finer spectral measurements than multispectral instruments, with often hundreds of spectral bands of narrow width being recorded, allowing a near continuous spectrum to be reconstructed for each pixel. This presents opportunities for more precise identification of biochemical and biophysical properties of the vegetation compared to when broadband multispectral sensors are used. The downside is the substantial increase in data volume and complexity.

Direct approaches to assess biodiversity using RS are based on analysis of dominant species over larger areas (Turner et al. 2003). These methods map the composition, abundance and distribution of individual species or assemblages and can be used to directly quantify habitats. Indirect approaches use remotely sensed data to measure environmental variables or indicators that are known or understood through biological principles to capture aspects of biodiversity (Duro et al. 2007). These include measures of: (i) the physical environment itself, such as climate and topography; (ii) vegetation production, productivity or function; (iii) habitat characteristics such as spatial arrangement and structure; and (iv) metrics of disturbance which can provide indirect measures of changes in biodiversity.

A wide range of in situ and remote sensing products [e.g., vegetation indices such as the Normalized Difference Vegetation Index (NDVI) and Foliage Projected Cover (FPC)] are beginning to be used for ecological monitoring in a variety of research projects and operational programs. Several satellite sensors [e.g., those on board of Landsat, Indian Remote Sensing Satellite (IRS) and SPOT satellites] have

been providing repeated global coverages for several decades. However, significant new opportunities are being presented with the increased availability of very high resolution images, hyperspectral data, Synthetic Aperture Radar, and LiDAR data. Their application has yet to be developed into routine and operational use in surveillance and monitoring of ecosystems, but soon will be.

#### **2.4.3.1 Ecosystem Extent and Distribution**

Trends in ecosystem extent and distribution are highly dependent on the scale of the evaluation being undertaken. For example, at a given scale, coastal wetlands may appear to be uninterrupted and uniform. However, at a more resolved scale, edges, patches, corridors associated with tidal creeks, and discontinuous distributions of species become evident. Forested and tree rich landscapes have a high connectivity for forest birds, but that may not be the case for carabid beetles and butterflies. Defining systems in terms of local organisation or dominant species facilitates discussion and analysis, but may also obscure the important linkages between systems across landscapes. It is therefore important to define the systems under consideration and the appropriate scale and resolution at which to observe and analyse them, before discussing trends in their extent and distribution.

Trends in the extent and distribution of ecological systems depend on the temporal and spatial scale of the assessment. Temporal changes occur naturally over long time scales, such as those associated with geological and climatological forces (e.g., glaciation). Change can also occur more quickly as a result of direct shifts in land use such as deforestation and urbanisation or the drainage of wetlands. Thus, trends can be the result of natural forces but may be accelerated by human pressure or exclusively due to human activities.

RS products have a high potential for mapping habitat extent and distribution maps at various scales. Hyperspatial (very high resolution) and hyperspectral sensors have greatly enhanced the possibilities of distinguishing related habitat types at very fine scales. The end-users can use such maps for estimating range and area of habitats, but they could also serve to define and update the sampling frame (the statistical ‘population’) of habitats for which field sample surveys are in place.

#### **2.4.3.2 Phenology**

Phenology is defined as the change in the life cycles of ecosystems and species through the seasons, for example the emergence of leaves or flowers. Phenology can be measured and analysed at different time scales, for example in hours to monitor water stress in crops and irrigation, days to manage plant stress from pests, quarters to monitor seasons, or years to understand seasonality and climate change. A convenient measure of plant phenology is the Normalised Difference Vegetation Index (NDVI)—an index which is available as a consistent data set for the entire Earth every 10 days at a resolution of 250 m (MODIS) and since 1982 for 8 km

imagery (NOAA AVHRR; see [http://phenology.cr.usgs.gov/ndvi\\_avhrr.php](http://phenology.cr.usgs.gov/ndvi_avhrr.php)). Other vegetation indices, such as the Enhanced Vegetation Index (EVI) avoid some the problems associated with NDVI (such as interferences caused by certain soils) and are possible to calculate using data from satellites launched after about 1995. Even better are direct measures of ecosystem function, such as the Fraction Absorbed Photosynthetic Radiation (FAPAR), which relates directly to Gross Primary Production, and is also a standard product of many modern Earth observation satellites.

Seasonal variations in any of the vegetation indices mentioned above can be used to track changes in vegetation phenology (Beck et al. 2007). ‘Hypertemporal’ imagery (i.e., observed every few days) can be parameterised using unsupervised classifiers and then used to map species distribution, such as a recent demonstration of mapping the extent of *Boswellia papyrifera* in Ethiopia. Such maps of species and biodiversity demonstrate a key advantage of long time series, an advantage of NDVI. Increasingly, landscapes are considered as gradients of particular traits, attributes and species rather than as discrete land cover classes. Treating the landscapes as gradients allows higher map accuracies to be achieved.

Vegetation indices have a spatial and a temporal dimension and so analysis and display of phenological processes can be challenging. For example, hypertemporal NDVI shows how vegetation greenness changes in time and with altitude. Remote sensing technology is being increasingly applied to studies of vegetation and ungulate habitats. For example, superimposing the movement data of radio-tracked giant pandas facilitates the visualisation of correlations between vegetation phenology and seasonal animal movement.

### 2.4.3.3 Connectivity and Fragmentation

Fragmentation is the process of breaking apart of previously uninterrupted patches of habitat and can have either negative or positive impacts on particular communities. Land and water development, land use and land use change are strongly fragmenting many landscapes and ecosystems e.g., by building highways through forests or damming rivers for hydro-electric power. The latter limits fish migration and separates essential parts of river ecosystems. Dams also reduce the populations of some species groups living in these ecosystems e.g., those that depend on running water, but increases habitat of others e.g., those that need still water. Fragmentation and the increasing length of edge habitat may force migrating species to find new ecological corridors, but may also allow new species (e.g., competitors, pathogens, weeds) to enter new areas. Regardless of specific impacts, fragmentation will in general result in smaller and more vulnerable ecosystems and in shifting the distribution of species.

Fragmentation can be measured through several metrics that can be calculated from RS data. The most simple is the Habitat Patch Density (HPD) that is defined as



the total number of areal elements within an area, for instance per km<sup>2</sup>. It is related to landscape grain and the composition of the landscape because the higher number of patches that are present a given area the higher is the landscape grain. The increase in HPD indicates an increase of the number of discrete elements in the landscapes and could lead to patch isolation when considering patches of the same habitat. According to meta-population theories, the increase in fragmentation and isolation may cause reductions in the flows of individuals and genes between habitat patches and can therefore threaten the viability of populations (Hanski 1998). The interpretation of HPD should be associated with the type of habitat, since the sensitivity to fragmentation and changes in connectivity associated with isolation, are dependent on constituent habitats and species.

Fragmentation can also be measured through Habitat Patch Size (HPS) that is defined as the average size of a patch in a given area. The HPS is linked to the number of patches within a given area. Although the link between HPD and HPS is not simple, in general if the number of patches within a given area increases there is a reduction in the average patch area. HPS is an indicator related to fragmentation since when a decrease in HPS is related to habitat shrinkage and could results in loss of core habitat, favour edges and decrease connectivity between patches. It has a negative impact on the abundance of habitat specialist species, particularly in forests. It would be interesting to differentiate the HPS by habitat types in order to follow time trends and comparisons between regions. Some animal species, including birds, mammals and reptiles prefers large habitat patches that provide sufficient area to provide them with all the resources needed. A decrease in HPS will often result in a reduction of biodiversity. At the landscape level the effect could however be counterbalanced by habitat diversity and connectivity especially for insects and other small mobile species.

## **2.5 Relating RS and in Situ Observations: LCCS and GHC**

In recent years work has been done to enable harmonisation between RS land cover and in situ habitat data. The monitoring of changes in land cover is important for the monitoring of changes in structural biodiversity. In many cases land use can be inferred from the land cover through virtue of its spatial configuration and context, e.g., a field of maize. Habitat maps can be derived from land cover maps based on RS data along with ancillary geographic information (e.g., soil maps) and other data derived from remote sensing data, e.g., Digital Elevation Models (Mücher 2011).

Where more than one system is used, the relationships between the components of these systems need to be made explicit (Scholes et al. 2012). Additionally, the harmonisation of land cover maps and habitat maps is very important, as habitats



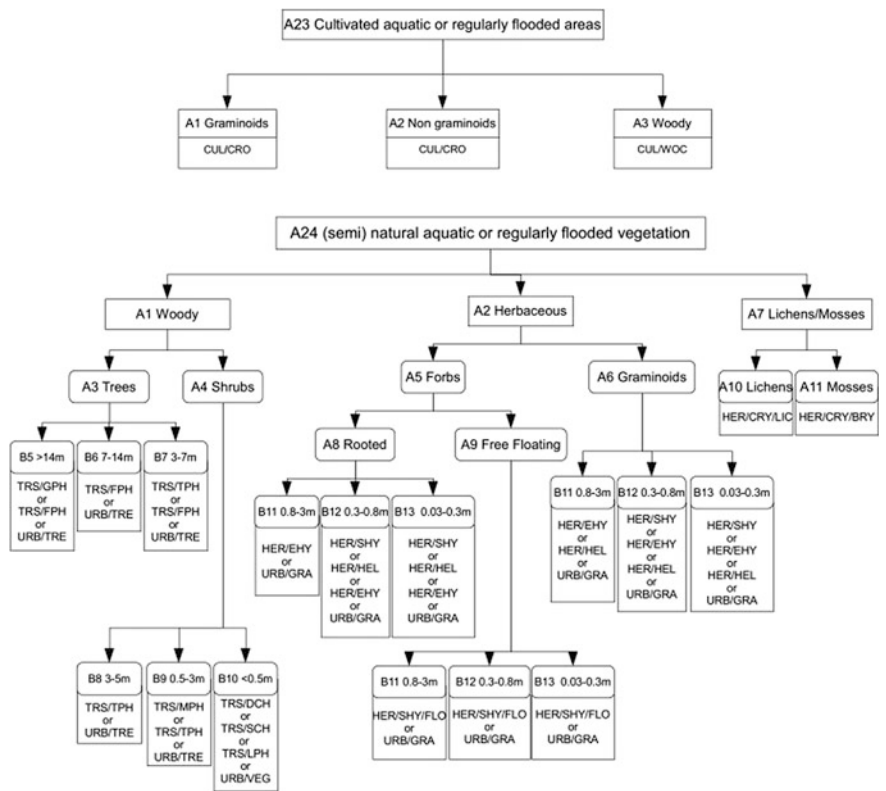
have strong associations with floristic and faunal taxa and are therefore considered significant as indicators of biodiversity (Bunce et al. 2013). It is a challenge to combine RS and in situ biodiversity observation systems to monitor changes in land cover and habitat reliably and to better understand the implications on habitat quality and the flora and fauna that it contains. Various initiatives have produced an increasing number of datasets with different classification schemes and mapping integrated yet.

To harmonise global ecosystems (or habitats as their spatial expression), use can be made of Plant Life Forms as first developed Raunkiaer (1934), elaborated by Küchler and Zonneveld (1988) and recently elaborated in the FAO-Land Cover Classification System (LCCS) for land cover interpretation of RS images (Jansen and Di Gregorio 2002), and in the GHCs (Bunce et al. 2008). Plant Life Forms are correlated with the main environmental gradient from the equator to the arctic and therefore can be used in both land cover and habitat mapping. Although LCCS and GHCs both use plant life forms as a basis, they were independently developed and therefore have small differences. Habitat classes are invariably related to land cover classes, but have more ecosystem-based characteristics. A translation system between GHCs and LCCS is important because this links land use as a driver of change and habitats as the spatially explicit representation of biodiversity.

LCCS has been used and proved valuable in land cover interpretation in Africa and Europe. The GHCs represent an important level of information on the status of biodiversity and habitats of good quality can be considered as a proxy for species occurrence. For instance, birds such as the bittern (*Botaurus stellaris*) can only be found in reed marshes and the European large blue butterfly (*Phengaris arion*) only in calcareous grasslands. Vegetation structure is central to both LCCS and the GHC classification and it therefore facilitates interaction between the GHC and LCCS taxonomies (Kosmidou et al. 2014). The main height categories of life forms are comparable between the two approaches with minor differences as shown in Table 2.2. As GHCs have in some cases a more detailed system, the translation between the two approaches requires in some cases ancillary data (Fig. 2.5).

**Table 2.2** Vegetation height definitions in the LCCS and GHC taxonomies

Height (m)	LCCS		GHC
>40	A12.A3.B5 Trees		Giga Phanerophytes, GPH Forest Phanerophytes, FPH
14–40			
7–14	A12.A3.B6 Trees		
5–7	A12.A3.B7 Trees		
3–5	A12.A3.B7 Trees	A12.A4.B8 Shrub	Tall Phanerophytes, TPH
2–3	A12.A4.B9 Shrub		
0.6–2			
0.3–0.6	A12.A4.B10 Shrub		Mid Phanerophytes, MPH
0.05–0.3			Low Phanerophytes, LPH
			Shrubby Chamaephytes, SCH
<0.05			Dwarf Chamaephytes, DCH



**Fig. 2.5** Relation table between the A23 and A24 LCCS categories and the corresponding GHC classes. *Source* Kosmidou et al. (2014)

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