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Operational models for the economic valuation of biodiversity services in forest ecosystems

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ABBREVIATIONS

Term	Explanation
CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
EBC	Ecology, Biodiversity and Conservation
EBV	Essential Biodiversity Variables
EC	European Commission
ES	Ecosystem Service
ESC	Ecosystem Capacity
ESTAT	Eurostat, Statistical office of the European Union
EU	European Union
EUNIS	European Nature Information System
EUSTAFOR	European State Forest Association
FAO	Food and Agriculture Organisation of the United Nations
FES	Forest Ecosystem Services
GIS	Geographic Information Systems
HD	Habitats Directive
InVEST	Integrated Valuation of Environmental Services and Tradeoffs
IUCN	International Union for Conservation of Nature
JRC	Joint Research Centre
LULC	Land Use Land Cover
MEA	Millennium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
MCPFE	Ministerial Conference on the Protection of Forests in Europe
NCA	Natural capital accounting
NEA	National Ecosystem Service Assessment
NFI	National Forest Inventory
NGO	Non-Governmental Organisation
NPP	Net Primary Production
SEEA	System of Environmental Economic Accounts

TEEB	The Economics of Ecosystems and Biodiversity
UN	United Nations
UNEP	United Nations Environment Programme

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FOREWORD

The main aims of the project BIOPROSPECT are to explore and document the bioprospects of forested protected areas and the ways of sustainable capitalization as a mean for their wise management and conservation, to encourage cooperation partnerships and networking among economic development planners and PA managers, to develop a cross-border bioprospect assessment methodological framework and economic valuation model in order to achieve outcomes which benefit both economic development and conservation.

BIOPROSPECT Work Package 3 aims to develop a tool box for the economic valuation and sustainable capitalization of biodiversity-ecosystem services. This will be achieved through the specific project objectives; to provide operational tools for the conservation of forest biodiversity through economic valuation and sustainable capitalization.

This report, (deliverable D3.1.3 under Task 3.5 in Work Package 3) approaches this objective by providing operational models for the economic valuation of biodiversity services in forest ecosystems.

The starting point of this report is an introduction to the conceptual framework of forest ecosystem services and biodiversity and a review of quantification as well economic valuation of biodiversity. The report presents forest biodiversity indicators and also targets to develop and analyze GIS forest services indicators

EXECUTIVE SUMMARY

Deliverable 3.1.3 (D3.1.3), under Task 3.5 in Work Package 3 (WP 3) - operational models for the economic valuation of biodiversity services in forest ecosystems, approaches the development biodiversity indicators for quantification and economic valuation of forest biodiversity. The report presents forest biodiversity indicators and also targets to develop and analyze GIS forest services indicators.

The report is structured in six main sections. The Introduction (Section 1) provides information about the concept under which the D3.1.3 is implemented, highlights the important of economic valuation of forest biodiversity.

Section 2 as the starting point in the analysis was to specify the concept of biodiversity, forest biodiversity and the connection with ecosystem services (ES).

Section 3 give the definition of ES indicator, analyses forest biodiversity indicators as practical way to characterize biodiversity and presents the Essential Biodiversity Variables (EBVs), as intermediate layer between raw data and indicators for study, reporting, and management of biodiversity change.

Section 4 illustrates the role of remote sensing and geographical information in quantification of biodiversity and forest services. Section 4 also presents a number of recent studies employing RS data and GIS to study biodiversity.

Section 5 refers to the Development of GIS forest services indicators and important landscape metrics in the field of biodiversity appropriate to describe the state of biodiversity concerning the relationships between landscape and biodiversity.

In section 6 economic methods and models of biodiversity valuation are shortly described. More over section 6 cites several case studies of economic valuation of biodiversity.

The report closes with section 7 where socioeconomic data over the designated Natura 2000 areas were gathered analyzed and mapped

ΕΚΤΕΤΑΜΕΝΗ ΕΛΛΗΝΙΚΗ ΠΕΡΙΛΗΨΗ

Το παραδοτέο D3.1.3, το οποίο ανήκει στο Πακέτο εργασίας (WP 3) - Επιχειρησιακά μοντέλα οικονομικής αξιολόγησης της βιοποικιλότητας σε δασικά οικοσυστήματα, προσεγγίζει την επιλογή ή/και σχεδιασμός των παραμέτρων και δεικτών για την οικονομική αποτίμηση της βιοποικιλότητας και των μετρήσιμων βιοφυσικών και παραγωγικών χαρακτηριστικών των δασικών οικοσυστημάτων

Η έκθεση διαρθρώνεται σε πέντε βασικά τμήματα. Η Εισαγωγή (Ενότητα 1) παρέχει πληροφορίες σχετικά με την έννοια στην οποία εφαρμόζεται το D3.1.3. όπου υπογραμμίζεται η ανάγκη οικονομικής αποτίμησης της δασικής βιοποικιλότητας.

Η ενότητα 2, ως αφετηρία της ανάλυσης προσδιορίζει την έννοια της βιοποικιλότητας, της δασικής βιοποικιλότητας και της σύνδεσής της με τις οικοσυστημικές υπηρεσίες (Ο.Υ)

Στην ενότητα 3 περιγράφεται η έννοια των δεικτών δασικής βιοποικιλότητας ως ένα πρακτικό εργαλείο για τον χαρακτηρισμό της βιοποικιλότητας. Ακόμα παρουσιάζονται οι Σημαντικές Μεταβλητές Βιοποικιλότητας -Essential Biodiversity Variables (EBVs), όπως αναπτύσσονται από τον GEO BON, ως απαραίτητες μετρήσεις για τη μελέτη και διαχείριση της βιοποικιλότητας

Η ενότητα 4 παρουσιάζει το ρόλο της τηλεπισκόπησης και των γεωγραφικών πληροφοριών (ΓΣΠ) στην ποσοτικοποίηση της βιοποικιλότητας και των δασικών υπηρεσιών. Στην ίδια ενότητα επίσης γίνεται αναφορά σε ορισμένες πρόσφατες μελέτες που χρησιμοποιούν δεδομένα τηλεπισκόπησης και των γεωγραφικών πληροφοριών για τη μελέτη της βιοποικιλότητας.

Η ενότητα 5 αναφέρεται στην ανάπτυξη δεικτών ΓΣΠ δασικών υπηρεσιών και σημαντικών μετρήσεων τοπίου για την περιγραφή της κατάστασης της βιοποικιλότητας.

Η ενότητα 6 όπου περιγράφονται σύντομα μέθοδοι και οικονομικά μοντέλα αποτίμησης της βιοποικιλότητας, όπως επίσης παραθέτονται μελέτες οικονομικής αποτίμησης της βιοποικιλότητας.

Η έκθεση κλείνει με την ενότητα 7 όπου συλλεχθέντα κοινωνικοοικονομικά δεδομένα για τις καθορισμένες περιοχές Natura 2000 αναλύθηκαν και παρουσιάστηκαν σε χάρτες.

1 INTRODUCTION

Biodiversity requires attention firstly due to a wide range of indirect benefits to humans and secondly, human activities have been contributing to unprecedented rates of biodiversity loss, which threaten the stability of ecosystems in terms of their provision of goods and services to humans (Dasgupta, 2001). There is a clear need to obtain information about the cause, type, and persistence of stress on biodiversity and the estimation of the respective impacts on human welfare (Nijkamp et al., 2008).

Environmental economics can inform conservation biologists and policy makers about why species are endangered, the opportunity costs of protection activities, and the economic incentives for conservation (Martín-López et al., 2008). The Convention of Biological Diversity's Conference of the Parties decision IV/10 acknowledges that "economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures" and encourages Parties, Governments and relevant organisations to "take into account economic, social, cultural and ethical valuation in the development of relevant incentive measures"(Secretariat of the Convention on Biological Diversity, 2005).

The economic valuation of natural resources, in general, and biodiversity, in particular, is among the most pressing and challenging issues confronting today's environmental economists. Biodiversity has both quantitative and qualitative characteristics. It is generally accepted that biodiversity cannot exclusively be expressed in numbers, as it also depends on the ecological structure of a whole area. Consequently, biodiversity tends to become a scarce economic good, for which however a proper pricing system does not exist (Nijkamp et al., 2008). Scientists argue that economic criteria need to be a part of the design and implementation of conservation policies (MEA, 2005). The combination or integration of the ecological and economic characteristics to assess and value biodiversity leads to an integrated framework (Nijkamp et al., 2008). Apart from the lack of a solid economic valuation mechanism for biological diversity, there is also a serious lack of reliable and up-to-date information and monitoring systems with a sufficient geo- graphical detail on biodiversity (Nijkamp et al., 2008). The imperative need for biodiversity protection, led to the adoption in 2002 of the 2010 Biodiversity Target and to considerable effort to identify and develop indicators at a global level. The generally poor level of information on biodiversity currently available, the policy response to the loss of biodiversity oblige EU Member States and signatories to the Convention on Biological diversity (CBD). The 2010 biodiversity target is the keystone of the Strategic Plan of the Convention on Biological Diversity (CBD) (Strand et al., 2007) which was adopted by Parties to the CBD. The convention on biological diversity (CBD) has recently established new targets towards 2020, the so-called Aichi targets, and updated proposed sets of indicators to quantitatively monitor the progress towards these targets.

Failing to meet the 2010 targets (Butchart et al. 2010), new indicators and an updated organization of all biodiversity indicators under 12 headline indicators were proposed (AHTEG 2011), meant to monitor the progress towards the achievement of the 20 Aichi Targets (CBD 2012) (Petrou et al., 2015). Upon demand by CBD, GEO BON attempted to assess the adequacy of global observation systems, mainly on information capacity, for monitoring biodiversity and the achievement of the Aichi targets (GEO BON 2011). (Pereira et al., 2012) suggest a set of candidate Essential Biodiversity Variables (EBVs).

2 BIODIVERSITY AND FOREST ECOSYSTEM SERVICES

2.1 Biodiversity

In recent years, the awareness has grown that biological diversity is of critical importance for the stability of the earth's ecosystem, as it forms the base for sustainable functions of natural systems (Nijkamp et al., 2008).

According to the Convention on Biological Diversity, '*Biological diversity' means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part. This includes diversity within species, between species and of ecosystems* (article 2 of the 1992 Convention on bi logical diversity). In more general terms, biodiversity conveys the biological richness of planet Earth (Steiger, 2005).

Biodiversity could be represented by of the three organisation level (Figure 1) (gene, species, habitat landscape) (Burkhard and Maes, 2017). At each level of biodiversity, three fundamental characteristics of biodiversity can be considered: composition, structure and function (Waldhardt and Otte, 2000). The variability across elements within levels of organization can be measured as richness (the number of elements), evenness (the equitability of elements), and heterogeneity (the disparity in element form and function) (Balvanera et al., 2014). Especially at the landscape level, composition and structure can be described by landscape metrics (Walz, 2011). Biodiversity also depends on geo-diversity, i.e., the variety of natural conditions, such as relief, soil characteristics and local climate, but in cultural landscapes also on the land use (Walz, 2011).

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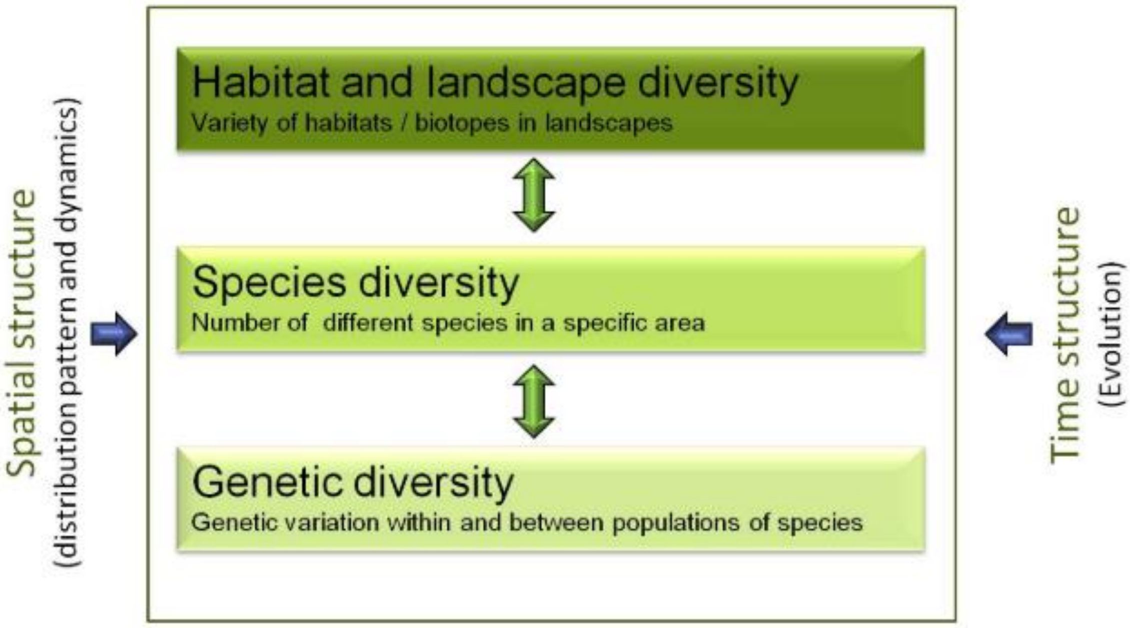

Figure 1Levels of biological diversity. Source: Walz, 2011

2.2 Biodiversity and ecosystem services

Biodiversity is our main capacity to generate ES and to ensure adaptation to environmental changes. Biodiversity is related to ecosystem services through a variety of mechanisms operating at different spatial scales (Burkhard and Maes, 2017). Figure 2 represents the essential components of the natural capital and the connection with ES and nature conservation. Biodiversity is linked to ecosystem services in three different ways (Figure 3): (i) as a regulator of the ecosystem functions that lead to the supply of provisioning, regulating services, (ii) as a provisioning service, (iii) as something that is appreciated in itself rather than for the benefits obtained from it. Source (Walters and Scholes, 2016)

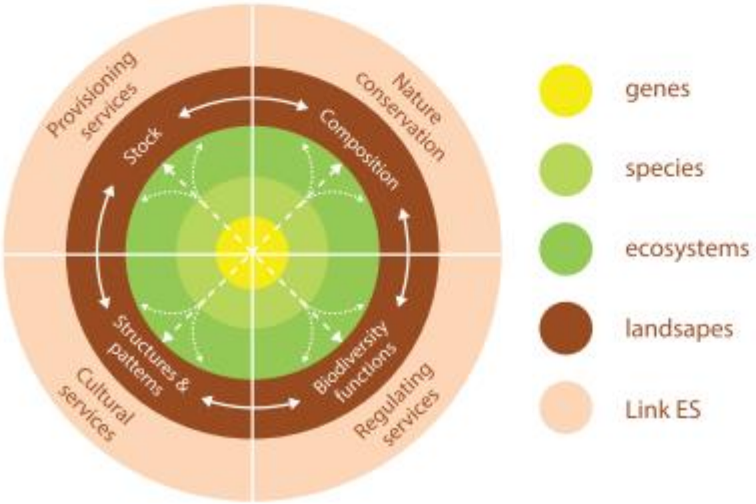

Figure 2 Four complementary perspectives of biodiversity, applicable to four organisation levels (gene, species, ecosystem & landscape) Source: Burkhard and Maes, 2017

Mace et al., 2012 propose that the confusion over the role that biodiversity plays in ecosystem services can be resolved by recognizing that different relations exist at the various levels of the ecosystem service.

i) Biodiversity can be a regulator of fundamental ecosystem processes. Biodiversity and ecosystem functioning underpins the delivery of all ecosystem services and should be accounted for in all decision-making related to the use of natural resources and areas (Vihervaara et al., 2015).

ii) a final ecosystem service itself, biological diversity at the level of genes and species contributes directly to some goods and their values

iii) or a good, biodiversity itself is the object valued by humans. Many components of biodiversity have cultural value, humans value places with a diversity of species, especially the more charismatic animals and plants, and retaining a full complement of wild species is important to many.

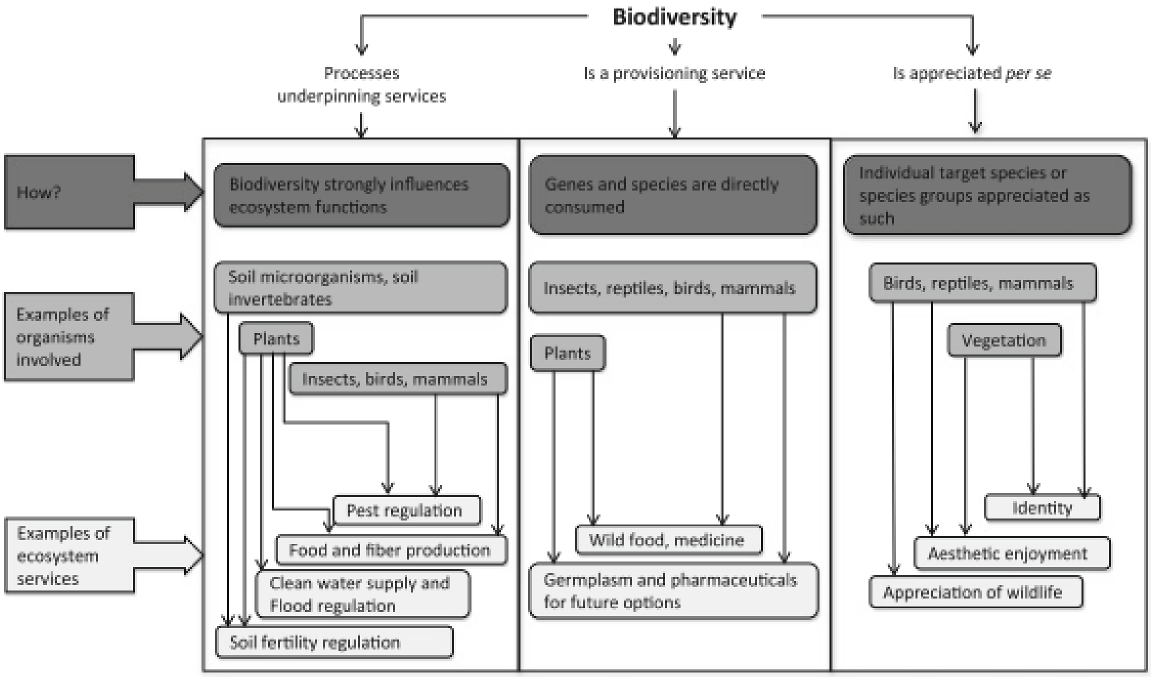

Figure 3. Biodiversity links to ecosystem services Source: Walters and Scholes, 2016

In principle, the supply of ecosystem services is mediated by each measure of biodiversity and at each level of organization (Balvanera et al., 2014). Harrison et al., (2014) conclude that regulating services were often associated with more Ecosystem Service Providers (ESP) and biodiversity attributes than other categories of ecosystem services compared to provisioning services.

2.3 Forest biodiversity

Globally, forests are vitally important for biodiversity (United Nations Environment Programme, 2001). Forests support approximately 80% of the world's terrestrial biodiversity (Balvanera et al., 2014) and it participates directly in production (wood, hunting, various forest amenities, etc.), in regulating production (resilience in the face of hazards and uncertainties) and in adapting to changes that are often complex and abrupt (Brahic and Terreaux, 2011).

In a forestry context, biodiversity most often refers to the biological diversity within forests at three levels: the genetic pool of living organisms, all the species of plants, animals and microbes, and all the habitats (Garcia et al., 2011). In the annex to COP 2 Decision II/9 (UNEP, 1996), the Conference of the Parties recognized that: Forest biological diversity results from evolutionary processes over thousands and even millions of years which, in themselves, are driven by ecological forces such as climate, fire, competition and disturbance. Furthermore, the diversity of forest ecosystems (in both physical and biological features) results in high levels of adaptation, a feature of forest ecosystems which is an integral component of their biological diversity. Within specific forest ecosystems, the maintenance of ecological processes is dependent upon the maintenance of their biological diversity. Loss of biological diversity within individual ecosystems can result in lower resilience.

Three main components of forest biodiversity (Figure 4) have been widely recognized (Spanos and Feest, 2007): composition, structure and function.

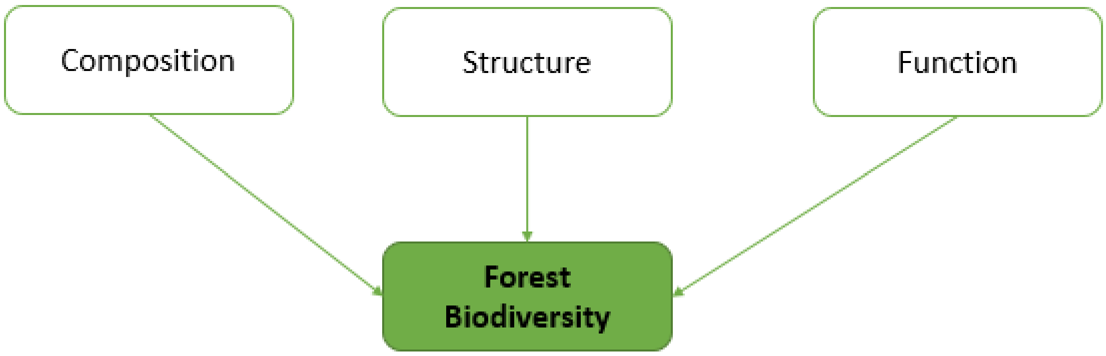

Figure 4 The three main components of forest biodiversity. Source: Spanos and Feest, 2007

Composition refers to the identity and variety of elements in a collection, and includes species lists and measures of species diversity and genetic diversity. Structure is the physical organization or pattern of a system, from habitat complexity as measured within communities, to the pattern or patches and other elements at the landscape level. Function involves ecological and evolutionary processes, including gene flow, disturbances, and nutrient cycling (Spanos and Feest, 2007).

3 ECOSYSTEM SERVICES INDICATORS

3.1 Forest services Indicators

Leaders in developing and developed countries—government, private sector, and civil society—need timely and targeted environmental indicators to understand the value and use of ecosystem goods and services, to analyze threats, and when combined with socio-economic indicators, assess the trade-offs at stake (Revenga, 2005).

ES indicators are information that efficiently communicates the characteristics and trends of ES, making it possible for policy-makers to understand the condition, trends and rate of change in ES (Burkhard and Maes, 2017). An indicator acts as a surrogate measure of more complex aspects of the reality being assessed, it can simplify the multivariate nature of the attribute being measured into a single value, thus allowing for spatial and temporal comparisons between values (Pereira et al., 2005). An indicator is commonly defined as ''something that provides a clue to a matter of larger significance or makes perceptible a trend or phenomenon that is not immediately detectable'' (Hammond et al. 1995). Strand et al., (2007) define an indicator as a measure used to determine the performance of functions, processes, and outcomes over time.

Indicators for provisioning ecosystem services delivered by forests were primarily drawn from national forest inventories and European forest data centers and relate to the production of timber. Regulating ecosystem services delivered by forest were poorly covered by available indicators. High quality indicators for cultural forest ecosystem services are not available which means that more work is needed to assess how forests contribute to this group of ecosystem services (Maes et al., 2016).

More information for forest service indicators is available in Deliverable D3.1.2 *Assessing the status and trends of forest services availability and distribution.*

3.2 Forest biodiversity indicators

Forest biodiversity is difficult task to be fully quantified, a practical way to characterize biodiversity is to use measures and develop biodiversity indicators (Granke et al., 2016). Biodiversity indicators are defined by the International Union for the Conservation of Nature as measures of biodiversity that help scientists, managers and politicians understand the condition of biodiversity and the factors that affect it (Dawson et al., 2016). A variety of different frameworks are available for the development and implementation of biodiversity indicators (Newton and Kapos, 2002). Different indicators employed in the assessment of the various biodiversity dimensions were recently proposed for approval by CBD, in order to measure the set of new global biodiversity targets for 2020 (Marques et al., 2013).

Forest biodiversity indicators are needed for many purposes (United Nations Environment Programme, 2001), including:

- ✓ reporting issues
- ✓ to identify priority areas and components of forest biodiversity
- ✓ to evaluate impacts of particular policies and decisions

According to (Newton and Kapos, 2002) biodiversity indicators can be divided into eight general groups

- ✓ forest area by type, and successional stage relative to land area;
- ✓ protected forest area by type, successional stage and protection category relative to total forest area;
- ✓ degree of fragmentation of forest types;
- ✓ rate of conversion of forest cover (by type) to other uses;
- ✓ area and percentage of forests affected by anthropogenic and natural disturbance;
- ✓ complexity and heterogeneity of forest structure;
- ✓ numbers of forest-dependent species;
- ✓ conservation status of forest dependent species.

Forest biodiversity indicators are generally selected following two main approaches (Granke et al., 2016):

1. counting the number of plant species in a given area (plant species richness)
2. by means of structural indicators based on key structural features (e.g. deadwood, variability in tree size, large trees) or quantified by indices of structural complexity.

Forest structures and/or tree species provide large numbers of habitats for different plant and animal species (Granke et al., 2016), measures related to trees and stand structure play a key role in the derivation of biodiversity indicators (Larsson, 2001). Continuing at the species level of biodiversity, trends in species populations are currently monitored globally with indicators such as the Red List Index (Butchart et al. 2007) and the Living Planet Index (Collen et al. 2009; Loh et al. 2005), with relevance to Aichi Target 12 (Thompson and Thompson, 2015). At the ecosystem structure level, land cover and forest cover change, and forest fragmentation are among the indicators that may be used to assess progress towards Aichi Targets 5 (Thompson and Thompson, 2015)

In general, indicators can be simple, based on single variables, or composite indicators, based on a combination of multiple variables, with the benefits of simplicity versus comprehensiveness (Strand et al., 2007). Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA) guidelines on designing national-level monitoring and indicators (CBD 2003) offer much practical guidance on the process of indicator development according to the process shown in Figure 5

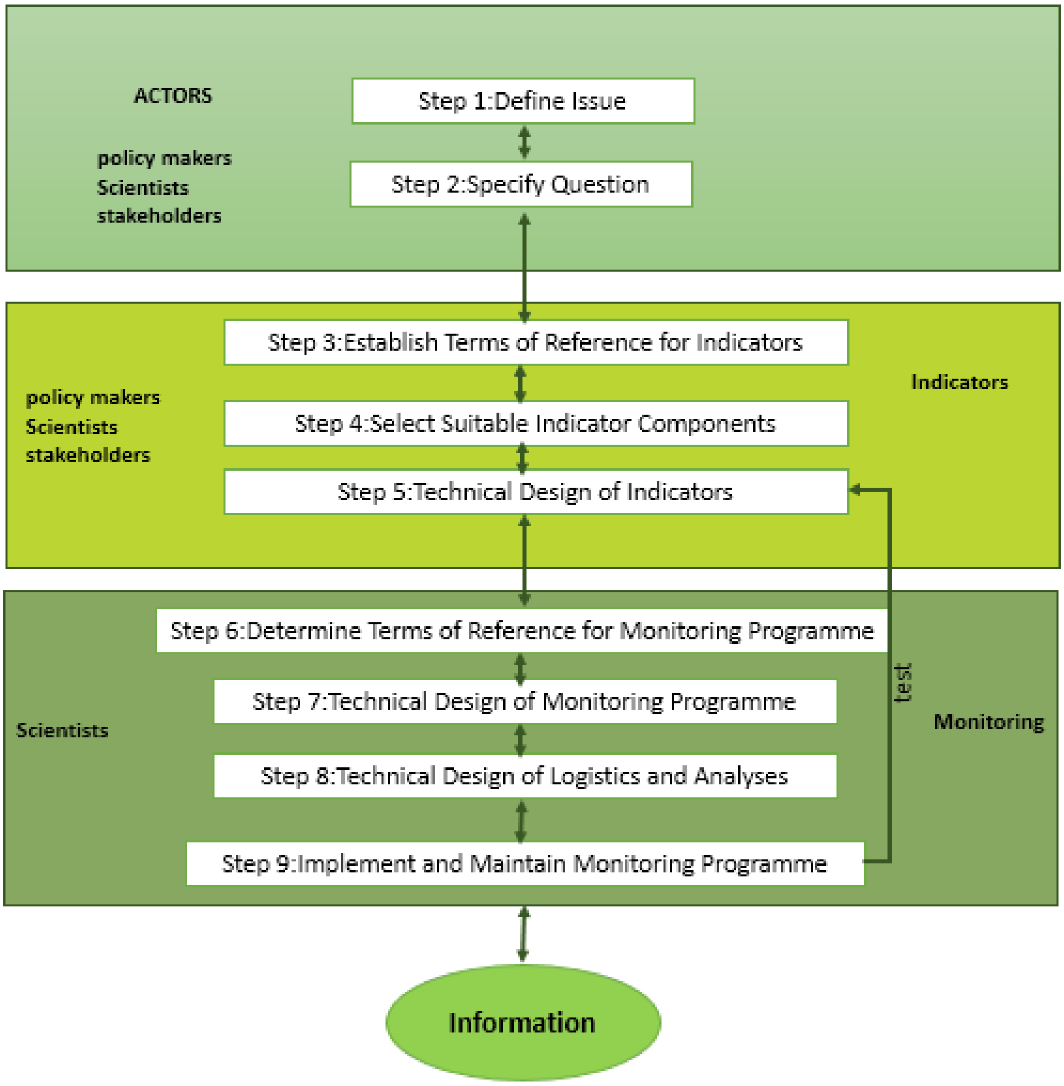

Figure 5 Steps in indicator selection and design. Source: Strand et al., 2007

3.3 Essential Biodiversity Variables

The Essential Biodiversity Variables (EBVs) (Pereira et al. 2013), under development by GEO BON, is defined as ‘a measurement required for study, reporting, and management of biodiversity change’(Walters and Scholes, 2016) and provide a critical use case for determining requirements for information systems. An EBV help in prioritizing by defining a minimum set of essential measurements to capture major dimensions of biodiversity change, complementary to one another and to other environmental change observation initiatives (Khare and Ghosh, 2016). EBV acts as an intermediate layer between raw data and indicators (Figure 6) whereby EBVs are the entities underpinning the generation of biodiversity indicators (Dawson et al., 2016)

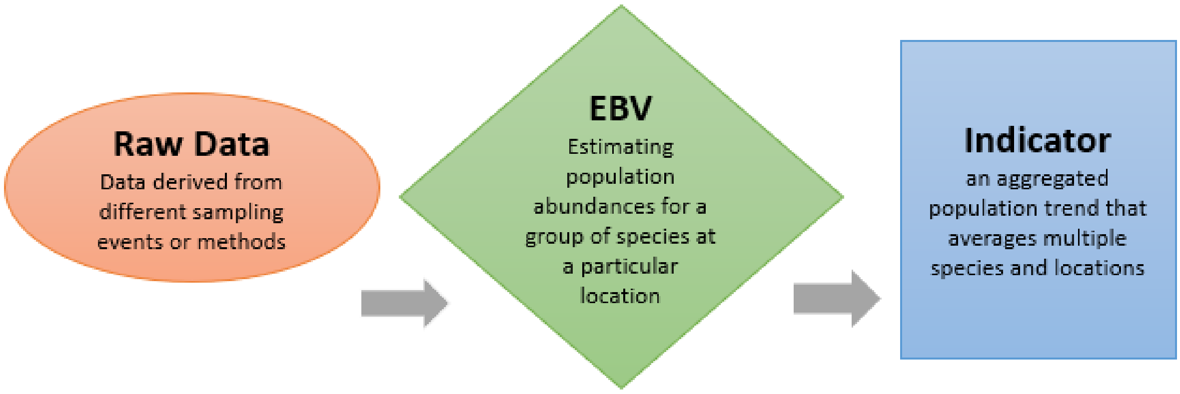

Figure 6 An EBV acts as an intermediate layer between raw data and indicators. Source: Walters and Scholes, 2016

GEO BON has identified six EBV classes; Genetic composition, species population, species traits, community composition, ecosystem structure, ecosystem function (Table 1).

Genetic composition

Genetic diversity is one of the essential variables of biodiversity, it is the fundamental aspect of the evolutionary process as it allows species to adapt to different environmental conditions by shapping their fitness (Marques et al., 2013).

Species populations

The species population dimension of biodiversity encompasses three different aspects a) species abundance, b) geographic distribution and c) risk of extinction. This is probably the best studied and documented dimension of biodiversity (Marques et al., 2013)

Species traits

Species traits concern the biological characteristics of a species, and are intimately connected with their niche. Also, traits determine the boundaries of the adaptative range of a species (Marques et al., 2013).

Community composition

Another essential dimension of biodiversity is community composition. Species that occupy the same geographical area at the same time establish relations that can be extremely sensitive to change; understanding these relations is fundamental to understand biodiversity (Marques et al., 2013).

Ecosystem structure

The structure of an ecosystem is established by the interaction between its biotic (communities of species) and abiotic components (Marques et al., 2013).

Ecosystem function

The function of an ecosystem concerns the web of flows, storages and regulations it can maintain (Marques et al., 2013)

Table 1 Metrics of Essential Biodiversity Variables (EBVs). Source: Marques et al., 2013

EVB	Indicadum	Headline indicator

Genetic composition Genotypes of selected species at representative locations	Allelic diversity and genetic differentiation	Domestic terrestrial animals, exploited species and wild relatives Cultivated plants, ex- situ crop collection and wild relatives
Species population Counts or presence surveys for groups of species over an extensive network of sites	Trends in abundance and distribution of selected species	Living Planet Index Global Wild Bird Index Average Species Abundance Alpha and Beta Richness Range Shifts Invasive and introduced Species
	Trends in threatened species and extinction risks	IUCN Red List Index
Species traits Timing of leaf coloration by remote sensing with in situ validation	Traits Diversity and Functional Richness	Phenology Multiple Trait Index Large Fish Indicator Functional Dispersion Functional Richness Community Weighted Mean Phylogenetic Diversity
Community composition Consistent multitaxa surveys and metagenomics at selected locations	Biotic Diversity	Taxonomic diversity Species per functional group Shannon- Wiener Index Simpson Index
Ecosystem structure Global or regional remote sensing of cover or biomass by height or depth Indicadum	Heterogeneity	Abiotic habitat structure Heterogeneity index Density of habitats
	Connectivity/ fragmentation of ecosystems	Forest fragmentation River fragmentation and flow regulation
	Species interaction	Connectance Food chain length Evenness of interactions Modularity Interaction length
Ecosystem function Nutrient output/input ratios at selected locations combined with remote sensing at regional scales	Exergy capture	Net Primary Production Leaf Area Index Exergy index
	Entropy production	Entropy Balance Carbon per year from respiration
	Storage capacity	Organic carbon and nitrogen in the soil Carbon and nitrogen in biomass
	Cycling and nutrient loss	Leaching of nutrients: Nitrogen and Phosphorus
	Biotic water flows	Transpiration per total evapotranspiration
	Metabolic efficiency	Respiration per biomass
	Water quality in aquatic ecosystems	Marine Trophic Index Water quality of freshwater ecosystems
	General	Ecosystem Functionality Index

4 QUANTIFICATION OF BIODIVERSITY AND FOREST SERVICES

There are various methods of measuring an indicator: running a model, direct measure or index lists. Determining what to measure and what method to use is directly related to the availability of data and the type of indicator (Marques et al., 2013).

Researchers developing models for mainstreaming ecosystem services assessments into the work of land- and water-use managers have suggested that the valuation of ecosystem services should include information resulting from both social and biophysical assessments (Cowling et al., 2008;Sherrouse et al., 2011). However, land use and management actions are better indicators of ESs than land cover (Ericksen et al. 2012; Koschke et al. 2013; van Oudenhoven et al. 2012).

4.1 Remote sensing

In general, the quantification of ecosystem services is a 2-fold indirect procedure. The remotely sensed information is used as a proxy for some kind of variable (e.g., biomass) which in turn is used as a proxy for the actual ecosystem service (e.g., carbon storage)(Ayanu et al., 2012).

Traditionally, ecologists map biodiversity and ecosystems based on in situ observations. 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 (Walters and Scholes, 2016).

Remote sensing can be used for measuring various environmental parameters, such as reflectance properties and three-dimensional (3D) structure of vegetation related to biodiversity (Saarinen et al., 2018). RS data are ideal for biodiversity monitoring because they can repeatedly provide systematically collected data over the entire earth (Thompson and Thompson, 2015). Although remote sensing has reduced the amount of expensive and laborious field assessments by improving the spatial resolution, quality of an assessment is still dependent on the availability and quality of field data.

Remotely sensed variable can be used to infer a range of ecosystem service changes through different model interpretations and interactions with other variables (Figure 7) (Walters and Scholes, 2016), e.g. supplies of provisioning services-production capacity of forests could be quantified via RS using biomass as an indicator. Regulatory services like air quality cannot be directly detected with remote sensing. However, the capacity of ecosystems to regulate air quality can be estimated through the assessment of their potential to remove or retain dust and reduce airborne pollutants (Ayanu et al., 2012). Ecosystems' capacity to influence climate can be estimated using carbon storage and sequestration as an indicator which is dependent on fluxes, emission, and aboveground storage of carbon (Ayanu et al., 2012). Moreover, RS is useful for estimating the capability of ecosystems to provide protection against extreme events such as storm, flood, and mass movements (Joyce et al., 2009). In general, the bio- physical dimension of ecosystem services has seen the most application of RS when estimating provisioning and regulation services. There have been few attempts to retrieve cultural services (Walters and Scholes, 2016).

The data collected from field plots is used for linking remote sensing measurements to the attributes of interests (Saarinen et al., 2018). Several models have been produced to illustrate this link of remote sensing data sets to forest parameters as forest structure through height, height variation, and density of the vegetation etc.

In Regression Models approach the quantification of ecosystem services is achieved by linking remotely sensed information to a limited number of in situ observations using semiempirical linear or nonlinear regression models (Ayanu et al., 2012).

Radiative Transfer Models allow a physically more direct derivation of biophysical parameters (Ayanu et al., 2012). Radiative transfer models are affected by our understanding of vegetation, land processes, and their interaction which increases uncertainty in the robustness and accuracy of ecosystem services quantification (Atzberger and Richter, 2012)

LULC has been widely used as a proxy for the quantification and mapping of ecosystem services RS provides useful data for LULC classification (Ayanu et al., 2012). Quantitative information about changes of land cover and land use, based on RS data, enable on landscape level the identification of the main pressures on biodiversity in the past (Granke et al., 2016).

Object Based Image Analysis (OBIA) is becoming a new paradigm in the field of RS due to its ability to handle complex information associated with new generation satellite datasets (Khare and Ghosh, 2016). A variety of studies have been carried using object based classification into the fields of Ecology, Biodiversity and Conservation (EBC). OBIA involves image segmentation, attribute selection, classification and the ability to link individual objects in hierarchy, actually it is based on the assumption that image objects provide a more appropriate scale to map environmental features and allows features with significant variations in their spectral reflectance signature to be mapped at specific scales (Blaschke, T., 2010)

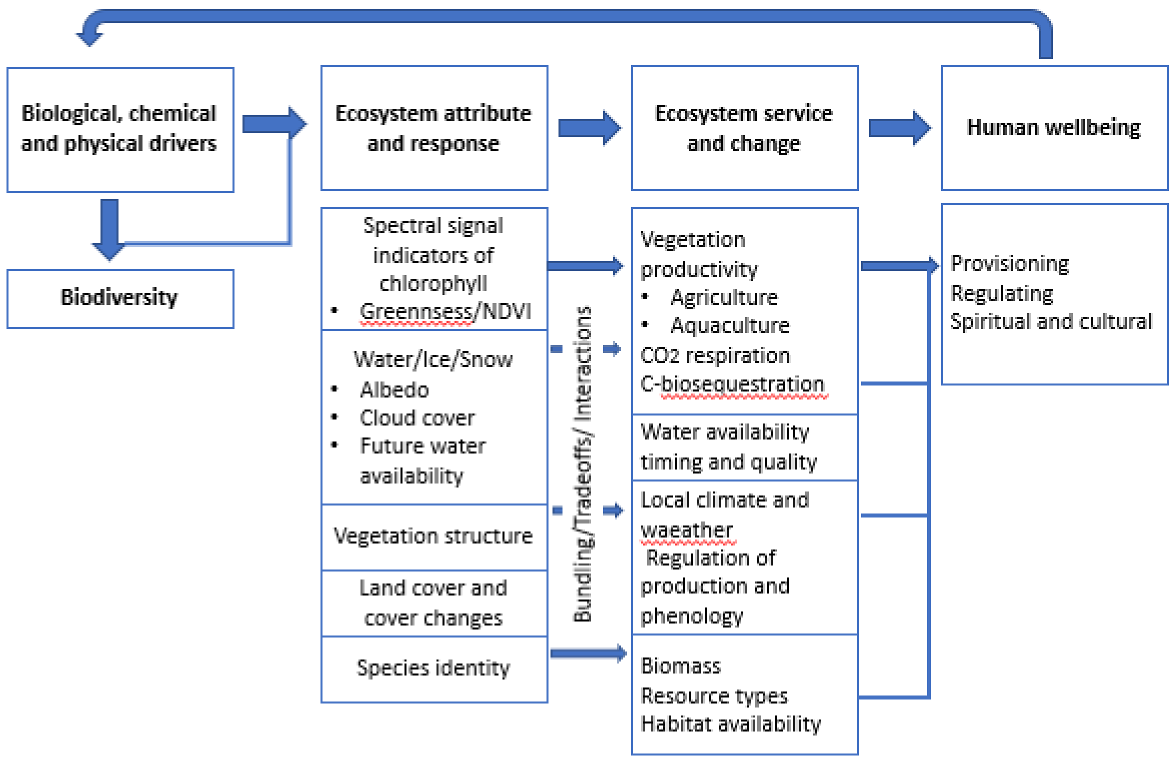

Figure 7 Remote sensing is used indirectly to evaluate changes in ecosystem services. Source: Walters and Scholes, 2016.

4.1.1 Remote sensing and biodiversity

Remotely sensed images do not represent biodiversity indicators per se. Rather, RS data form the raw inputs from which indicators can be constructed. For example, the signal to remote sensors can be associated with a particular vegetation cover type (such as forests). A change in the signal from one time period to another might indicate a change in vegetation cover and the habitat that is associated with that cover. Validation with ground truth or by high resolution data is necessary to confirm remote sensing observations. Data manipulation within a GIS environment can help produce the maps and statistics needed to create an indicator that can be understood by decision makers and the general public (Strand et al., 2007).

RS may be used to map habitat types or land cover across a landscape and to map the impacts of natural and human-caused processes causing fragmentation. Alternatively, the causes of fragmentation (e.g., roads, agriculture, or fire) can be mapped, and then the reciprocal of these areas can be classified as intact patches of habitat (Strand et al., 2007).

Table 2 identifies the CBD headline indicators that may be extracted through RS data, and the mainly associated Aichi targets.

Table 2 Aichi targets that can be monitored through RS data, and the associated CBD headline indicators (AHTEG 2011) and EBV classes (Pereira et al. 2013). Source: Petrou et al., 2015

Aichi targets	CBD headline indicators	EBV classes
(4) Sustainable production and consumption	(4) Pressures practices, (5) pressures various	Species populations
(5) Reduction of habitat loss, fragmentation and degradation	(1) Extent, (4) pressures practices, (5) pressures various	Species populations ecosystem function ecosystem structure
(6) Sustainable exploitation of marine resources	(4) Pressures practices	Species populations
(7) Sustainable management of agriculture, aquaculture and forestry areas	(4) Pressures practices	Species populations ecosystem structure
(8) Pollution reduction	(5) Pressures various	Species populations community composition ecosystem function
(9) Invasive alien species control	(2) Species, (5) pressures various	Species populations
(10) Protection of vulnerable ecosystems	(5) Pressures various	Species populations community composition ecosystem structure
(11) Conservation and protection of important areas	(11) Protected areas	Species populations ecosystem structure
(12) Preventing extinction of threatened species	(2) Species	Species populations
(14) Safeguarding ecosystems with essential services	(6) Services, (11) protected areas	Species populations community composition ecosystem function ecosystem structure
(15) Enhancing ecosystem resilience	(6) Services, (11) protected areas	Species populations species traits ecosystem structure

The index numbers of the Aichi targets and CBD indicators used in their definition documents are given in parenthesis. Abbreviations of CBD headline indicators: (1) Extent trends in extent, condition, and vulnerability of ecosystems, biomes, and habitats, (2) Species trends in abundance, distribution, and extinction risk of species, (4) Pressures practices trends in pressures from unsustainable agriculture, forestry, fisheries, and aquaculture, (5) Pressures various trends in pressures from habitat conversion, pollution, invasive species, climate change, overexploitation, and underlying drivers, (6) Services trends in distribution, condition, and sustainability of ecosystem services for equitable human well-being, (11) Protected areas trends in coverage, condition, representativeness, and effectiveness of protected areas and other area-based approaches

Table 3 presents the relationship among EBV and RS measurements

Table 3 Relationship among Essential Biodiversity Variables and Remote Sensing Measurement Scales. Source: Khare and Ghosh, 2016

EBV (Ecosystem Structure)	**Spatial Resolution satellite imagery with type of measurement scales (including available remote sensing sensors)**	**Relevance and related information for biodiversity**
Temporal phenology metrics	Low/coarser spatial resolution (Global Scale) (MODIS, AVHRR etc.)	Phenology types, Forest / Non Forest, Deforestation and Biomass burning.
Habitat Structure, Ecosystem extent and fragmentation	Medium spatial resolution (Regional Scale) (Landsat, IRS, SPOT etc.)	Forest type distribution and agricultural expansion
Habitat types and structures, and Ecosystem composition by functional type	High spatial resolution (Local scale) (IKONOS, QuickBird, Rapid Eye historic GeoEye, WorldView-2 etc.)	Species-level distribution, canopy diameters, stand-level analysis, individual tree detection, to differentiate species at a finer scale.
Habitat types and structures	Active remote sensing data	Habitat degradation monitoring by generation of 3D structures.

4.2 Geographic information systems

New technologies like RS and a geographic information system (GIS), facilitate the development of indicators of ecosystem condition and change, which can be used in management decisions and to establish long-term monitoring programs (Revenga, 2005). The most widely used definition of GIS is 'a computer-based system that captures, stores, manages, analyses, and displays georeferenced data (geographic data)'(Salem, 2003).

The use of a GIS for conducting integrated analyses of social and environmental data in a variety of contexts is well-documented (Sherrouse et al., 2011).

GIS allows for the combination of physical, biological, and socio-economic data to analyze ecosystem condition and change, making linkages between change and impacts possible (Revenga, 2005). GIS can be used to visualize ecosystem services distribution, to compare the distributions of multiple ecosystem services with drivers of change and other social-ecological parameters, and to model how changes in land use or land cover, land management, ecosystem and climatic conditions, and human populations affect ecosystem service provision and the value and use of services (Kareiva et al., 2011).

The proliferation of freely available satellite imagery and associated databases allows for a GIS-analysis of ecosystem services in areas of the world where few other forms of data are available (Nemec and Raudsepp-Hearne, 2013). Land-cover data, often manipulated or subject to modeling in a GIS, is the most common input for ES modeling (Andrew et al., 2015).

According to (Nemec and Raudsepp-Hearne, 2013), approaches to estimating ecosystem service values using GIS include:

(1) the development of 'static' estimates, or data-driven values that present a snapshot of current or past ecosystem services across a landscape;

(2) the development of ecosystem service models that can be used to analyze how changes in landscapes impact the provision of ecosystem services and benefits; and

(3) the development of models and approaches that emphasize social preferences and priority-setting for ecosystem service management.

Several programs have been developed recently to model multiple ecosystem services in a variety of systems. InVEST has possibly been the most widely used GIS software tool for mapping ecosystem services, and has been applied in decision-support processes in a diversity of geographic contexts (Nelson and Daily 2010). ARIES (ARtificial Intelligence for Ecosystem Services), an open-source GIS application that has recently been developed for applying the benefits-transfer approach (Nelson and Daily 2010).

Some more information for GIS and GIS applications as approach to quantify ES is available in Deliverable D3.1.2 *Assessing the status and trends of forest services availability and distribution.*

4.2.1 Geographic information systems and biodiversity

Information is needed to develop model strategies for biodiversity conservation strategy and users require biodiversity data on the context within they need to focus. Geographical information system (GIS) has provided, especially through remote sensing, a range of data on environmental properties as well as techniques to explore and use data to further understanding of biodiversity and aid its conservation (Foody, 2008). These data will be in the form of text documents, tabular databases, spatial databases (locations), image files (satellite images), and so on, and will include topographic, environmental, species, administrative, socioeconomic and other themes. The role of GIS is to integrate all these forms of data for assessment and monitoring purposes (Salem, 2003) as well to analyze potential and current spatial distribution of target species, measuring biodiversity, monitoring biodiversity patterns and identifying priorities for conservation and management (Krigas et al., 2012). GIS also can be used to offer a reliable, quantitative and qualitative description of the in situ habitat conditions preferred and/or tolerated by different target plant species in the wild (Krigas et al., 2012). Moreover, GIS has been widely used to study how landscape patterns impact on biogeographical variables and biodiversity (Figure 8). For example, issues such as the effect of road networks on accessible habitats and humans on habitat quality can be modelled with basic GIS tools (Foody, 2008) Flexible GIS-based tools have also been developed to exploit static information of botanical collections in an attempt to evaluate species distributional ranges (Krigas et al., 2012).

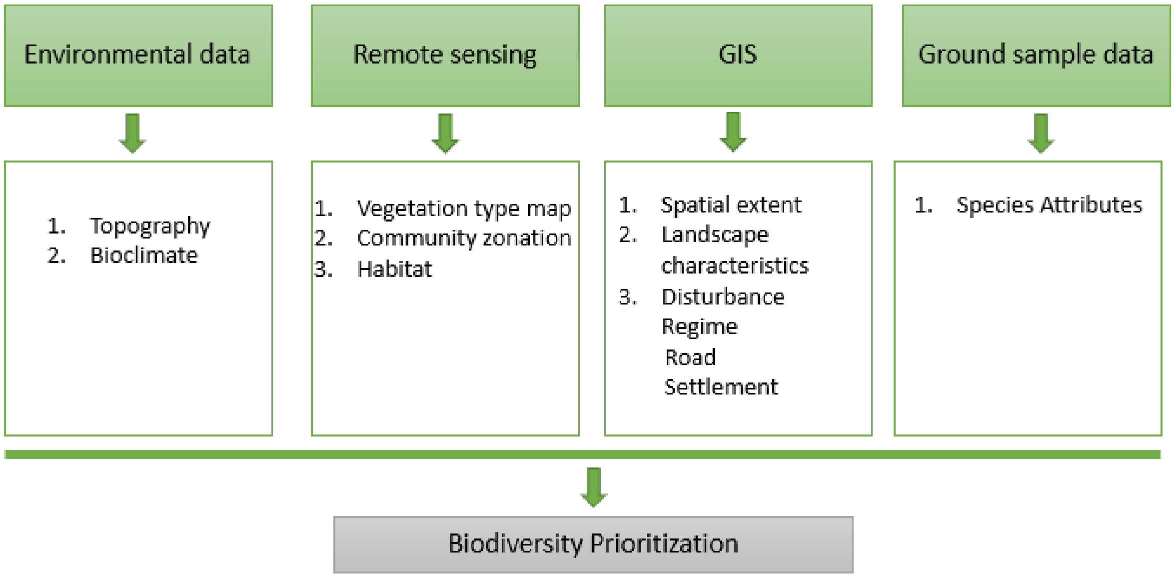

Figure 8 Approach for Biodiversity characterization Source :Roy and Saran, 2004

4.3 Case studies

A number of recent studies are successfully employing RS data and GIS to study biodiversity, either through direct monitoring or through proxy variables.

Saarinen et al., (2018) study estimates plot-level biodiversity indicators in boreal forests in southern Finland with hyperspectral imagery and photogrammetric point clouds from a UAV. Structural metrics from the photogrammetric point clouds were used together with either spectral features or vegetation indices derived from hyperspectral imagery. Biodiversity indicators that are used in this study include variability in both species and structural attributes; Species richness, amount of dead wood, structural heterogeneity, successional stage, amount of large deciduous trees.

Bagaram et al., (2018) highlights that UAV remote sensing can potentially provide covariate surfaces of variables of interest for forest biodiversity monitoring, which are conventionally collected in forest inventory plots.

Estes et al., (2010) used principal components analysis (PCA) and an additive SCI method developed for forest ecology (calculated by rescaling and summing representative structural variables) to summarize 13 microhabitat-scale vegetation structure attributes describing the rare mountain bongo antelope's habitat in Kenya's Aberdare mountains. Regression models based on variables derived from ASTER imagery processed with spectral mixture and texture analysis, digital elevation model and rainfall data.

Vihervaara et al., (2015) study quantify spatially-explicit biodiversity indicators for ecosystem assessments and evaluate the capacity of those indicators to describe biodiversity-ecosystem service relationship in the Lake Vanajavesi in southern Finland. They use Airborne laser scanning data for ecosystem structure and bird observation data.

Vaglio Laurin et al., (2014) use Airborne hyperspectral imagery to predict the alpha diversity of upper canopy trees in a West African forest. They conclude that airborne hyperspectral sensing can be very effective at mapping canopy tree diversity, because its high spatial resolution allows within-plot heterogeneity in reflectance to be characterized, making it an effective tool for monitoring forest biodiversity over large geographic scales.

Kumar et al., (2015) research incorporates field-based surveys along with remote sensing technologies LISS IV Sensor, using a regression model to estimate and recognize different species diversity in Sariska Tiger Reserve, in Alwar District, Rajasthan State, India.

Madonsela et al., (2017) estimate species diversity in the savannah woodland in southern Africa, using Landsat-8 Operational Land Imager dataset. Utilizing the entire spectral information in the Landsat-8 data enhanced the ability to estimate tree species diversity. They conclude that the significant relationship observed between remotely sensed variables and tree species diversity measures confirms the utility of Landsat image for practical application in conservation, particularly as a screening tool to identify biodiversity hotspots.

Kampouri et al., (2018) use Simpson's Diversity Index, derived through the combined summer-winter Sentinel-2 image classification, to map forest tree biodiversity. The results of Object Based Image Analysis (OBIA) showed great promise in identifying and monitoring biodiversity hotspots, disturbance and the effects of changes in management practices and climate change for small study areas.

Rocchini, (2009) applied a simple multi-date NDVI based Mahalanobis distance measure (called eco-climatic distance) for tree biodiversity and ecosystem services at two nested scales for the Western Ghats Biodiversity hotspot. His study further demonstrated that the RS based metric is a good approximate surrogate for various components of biodiversity at broad to very broad scales.

Roy et al. (2013) used a moving window approach to identify potential areas of forest fragmentation in Indian landscape to assess the impact of anthropogenic pressures and cultural practices on forest fragmentation that provides critical inputs for prioritization and conservation of forests and associated biodiversity.

Haas and Ban, (2018) evaluate the contribution of Sentinel-2A (S-2A) data to map forest urban habitat for biodiverity and to investigate spatial ecosystem service characteristics with landscape metrics in Beijing, China.

Onaindia et al., (2013) designed a GIS based approach with the aim of proposing criteria for conservation plans that would include ecosystem services and biodiversity, to estimate and map the value of the biodiversity and ecosystem services, in northern Spain.

Salata et al., (2017) mapped habitat quality in Lombardy (northwest Italy) using the InVEST (Integrated Valuation of Ecosystem Services and Tradeoff) model.

Chen et al., (2009) designed a GIS-based approach to spatially estimate direct use value of ecosystem services by linking GIS, geospatial data, biophysical data and socio–economic data, and to map results for a case study of the Tiantai County in southeast China. The approach highlights the use of GIS to collect data, perform spatial analysis, and map economic values of ecosystem services.

Jones et al., (1997) described method for applying geographical information systems (GIS) to exploring biodiversity in the wild relatives of crop species throughout Latin America and produce maps indicating areas with `bean-favouring' climates.

Debinski et al., (1999) used remotely sensed data and GIS to categorize habitats, then determined the relationship between remotely sensed habitat categorizations and species distribution patterns. in the Greater Yellowstone Ecosystem, USA.

Gupta and Sharma, (2012) study spatial distribution of plant biodiversity in Himachal Pradesh , in India. They apply remote sensing and GIS technology to produce information data on thematic maps, and to delineate land use types preparing a database of vegetation inventory under different land uses.

5 DEVELOPMENT OF GIS FOREST SERVICES INDICATORS

5.1 GIS Indicators

Many different types of data (polygons, vectors, raster) can be integrated into a GIS (Figure 9). When these layers are drawn on top of one another, important information's about spatial trends and relationships can be revealed.

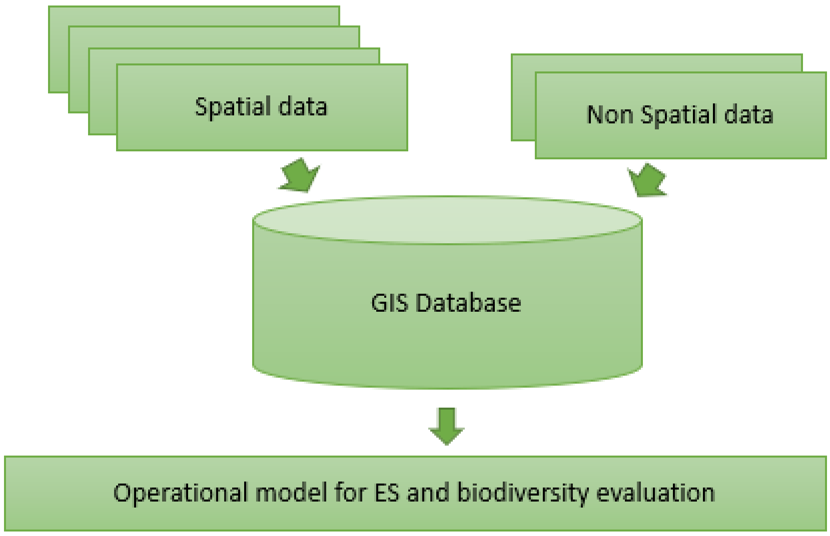

Figure 9 GIS operational models

Land use/land cover

The most widely used type of information in Ecosystem Service (ES) assessments is Land Use Land Cover (LULC) maps, where categorical spatial data mapping the distribution of vegetation types and other surface features of a study area (Andrew et al. 2015). Some human activities conducted in an area (land use) may be inferred. LULC data relied on to produce the spatially distributed biophysical parameter values needed for production function models (e.g., many of the InVEST models). LULC products provide abundant, detailed information relevant to many environmental patterns and processes (Andrew et al. 2015).

Topography

Elevation and topographic variables derived from digital elevation models (DEMs) feature prominently in models of hydrological services (Andrew et al. 2015). Slope surfaces derived from DEMs are frequently used to model erosion control and sediment regulation (Andrew et al. 2015). Models of tourism and aesthetic values also tend to enlist topographic surfaces, specifically, viewshed models (Andrew et al. 2015). Topography provides inputs to empirical models of a range of other services as greenhouse gas mitigation (Jenkins et al. 2010), timber production. Roughness of land surfaces adjacent to roads effects capacity to regulate the concentration in the air of the pollutants affecting human health and the quality of urban life (Barnes et al. 2014).

Road layers

Digitized road networks used to indicate access/ use of a service (e.g., recreation: Bateman, Lovett, and Brainard 1999; flood protection: Nedkov and Burkhard 2012, as well as potential environmental degradation reducing service provisioning (e.g., scenic views: Bagstad et al. 2012; recreation: Lautenbach et al. 2011).

Landscape metrics

Landscape metrics are tools which can be used to bridge the methodological gap between landscape structure and ES provision (Burkhard and Maes 2017). Spatial characteristics have implications on the performance of biodiversity and several ES, could quantified with landscape metrics (Haas and Ban 2018)

Fragmentation

Habitat fragmentation is the division of ecosystems or habitats into smaller, less connected patches (Strand et al. 2007). Lehmkuhl and Ruggiero (1991) have defined as fragmentation as the "creation of a complex mosaic of spatial and successional habitats from formerly contiguous habitat".

In the context of biodiversity monitoring, fragmentation is the result of anthropogenic acts such as logging, agricultural development, urban development, and infrastructure development (e.g., roads, agriculture, or fire) (Strand et al. 2007).

Vogt et al. (2007) developed an improved method for classifying forest fragmentation. The Landscape Fragmentation Tool v2.0 (LFA tool) uses (Figure. 10) an equivalent procedure that takes advantage of the capabilities of ArcGIS. The LFA tool is a python script that runs in ArcToolbox.

Four classes of forest are identified – in terms of the type of fragmentation present:

- Core – interior forest pixels that are not degraded from "edge effects".
- Perforated – forest along the inside edge of an small forest perforation.
- Edge – forest along the outside edge of a forest patch.
- Patch – small fragments of forest that are entirely degraded by "edge effects".

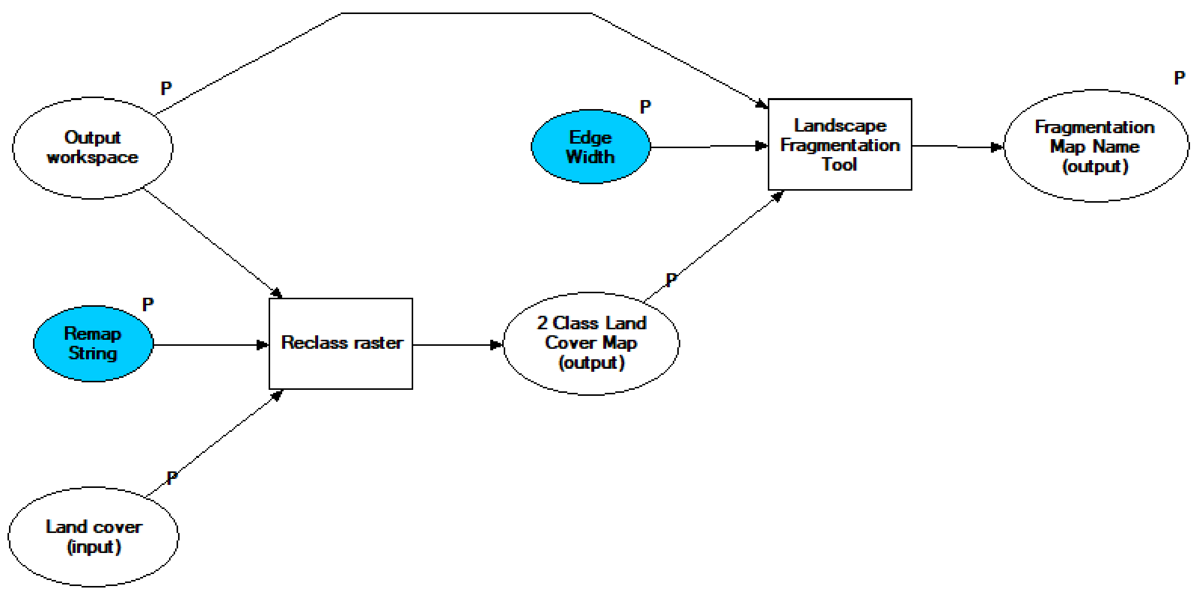

Figure 10 The Landscape Fragmentation Tool v2.0 of ArcGIS

Human footprint

Human footprint data as road layers, can be used to indicate use of ESs as well as degradation to ecosystems and ESs. Data products mapping anthropogenic impacts, typically by integrating the distribution of human settlements and accessibility (i.e., transport infrastructure, navigable rivers) (Andrew et al. 2015)

Hydrological parameters

Hydrological data may be derived from observations (e.g., gauging stations) or from model outputs. Although some are published data products, others are described in the gray literature and it can be difficult to ascertain how they were created and using what sources of information. These datasets are available variously in gridded format, associated with watershed polygons, or as point measurements (Andrew et al. 2015).

Waterbodies

Waterbodies location and density of surface water, including lakes, rivers, and streams used to inform water and water-constituent routing in hydrologic service models (Bagstad et al. 2011)

Soil maps

Soils are essential components of the earth system and play important direct and indirect roles in the provisioning of many ESs (Robinson et al. 2013). Categorical soil maps are the usual source of spatial soil data for ES assessment (Andrew et al. 2015).

Spectral indices

A spectral measure of “greenness”, related to the amount and activity of vegetation is the Normalized Difference Vegetation Index (NDVI). NDVI is used as an indicator of ESs related to plant production, carbon sequestration: Su et al. 2012; erosion control: Fu et al. 2011; forage production: Malmstrom et al. 2009; plant phenology: Walters and Scholes, 2016). The application of remote sensing in biodiversity estimation has largely relied on the NDVI (Madonsela et al. 2017) which supported by a large number of satellite and airborne instruments.

Environmental data

Environmental data may be used in assessing the relative biodiversity of the area, not because of interest in environmental variation per se, but because environmental (habitat or ecosystem) variation indicates species diversity. Species ranges and richness are often correlated with the habitat factors, and thus, both species ranges and habitat factors can be predicted from one another. Sometimes these two variables are combined into synthetic maps of ecoregions at the biogeographic scale (Salem 2003). Climate is generally regarded as the dominant control over the potential range of taxa. The bioclimatic factors, such as absolute minimum temperature and annual temperature range conditions during critical phases of a species life cycle (phenological stages), are limiting factors to species' ranges (Salem 2003)

Figures 11-21 present some examples of how GIS indicators quantified and map ecosystem services

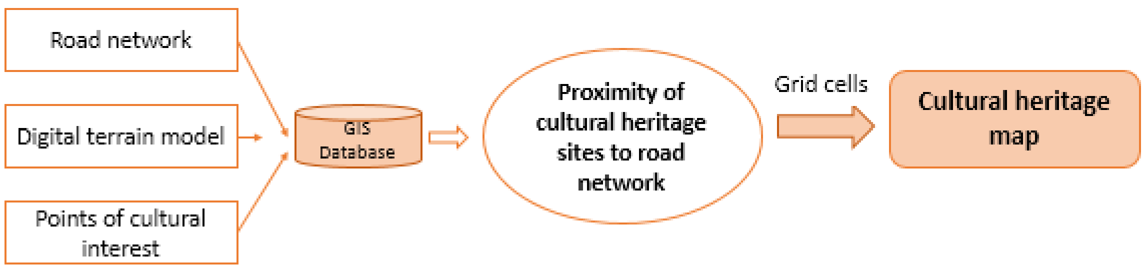

Figure 11 Cultural service - Cultural heritage

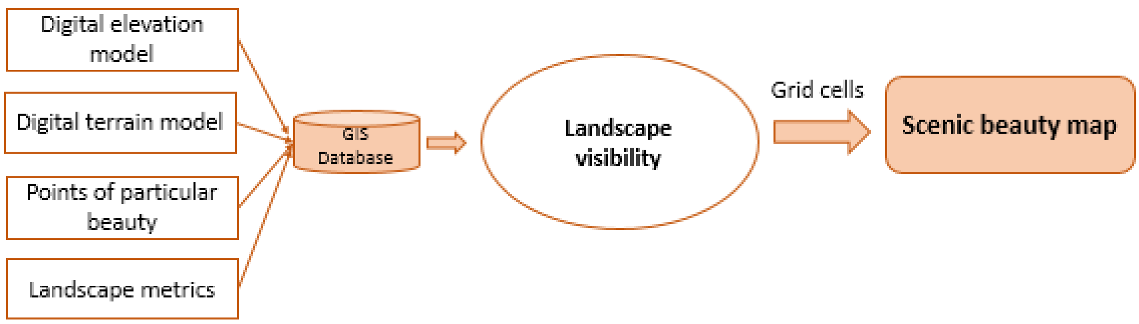

Figure 12 Cultural service -Scenic beauty

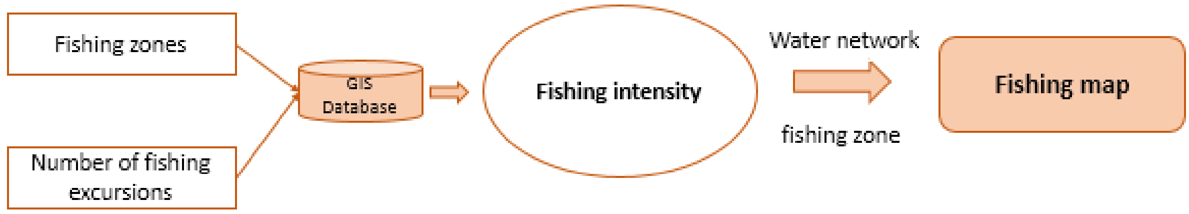

Figure 13 Cultural service – fishing map

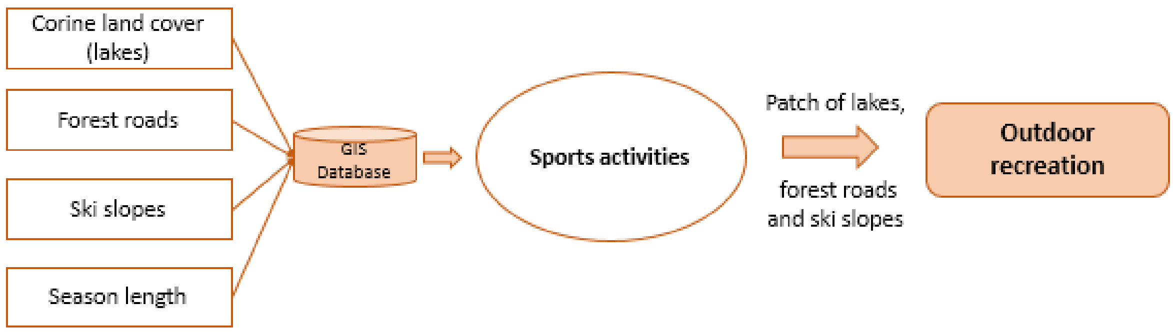

Figure 14 Cultural service – Outdoor recreation

Figure 15 Regulative service - Flood prevention capacity

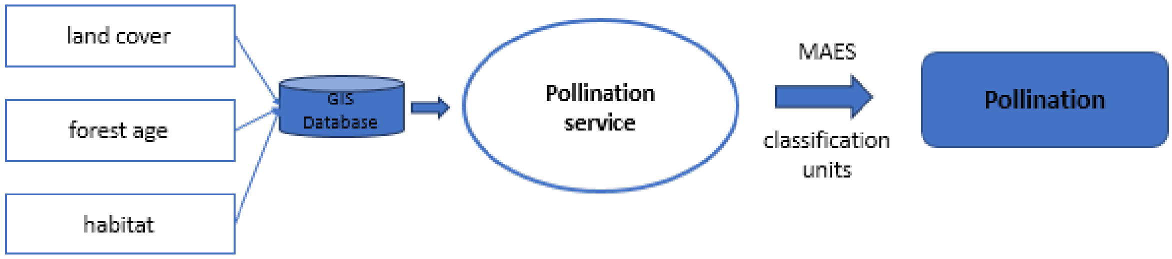

Figure 16 Regulating service – Pollination

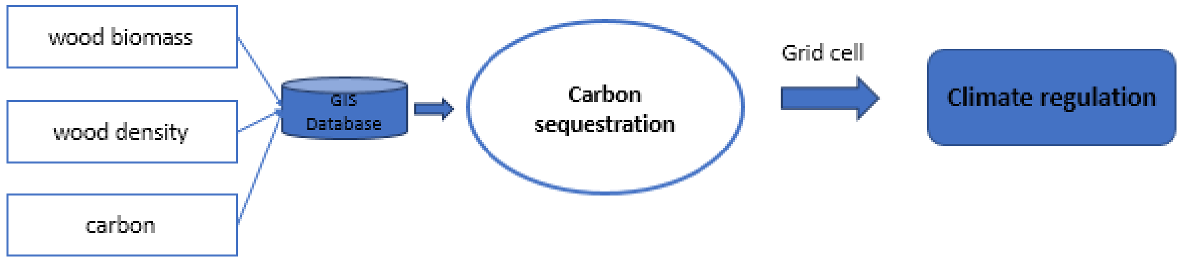

Figure 17 Regulating service – Climate regulation

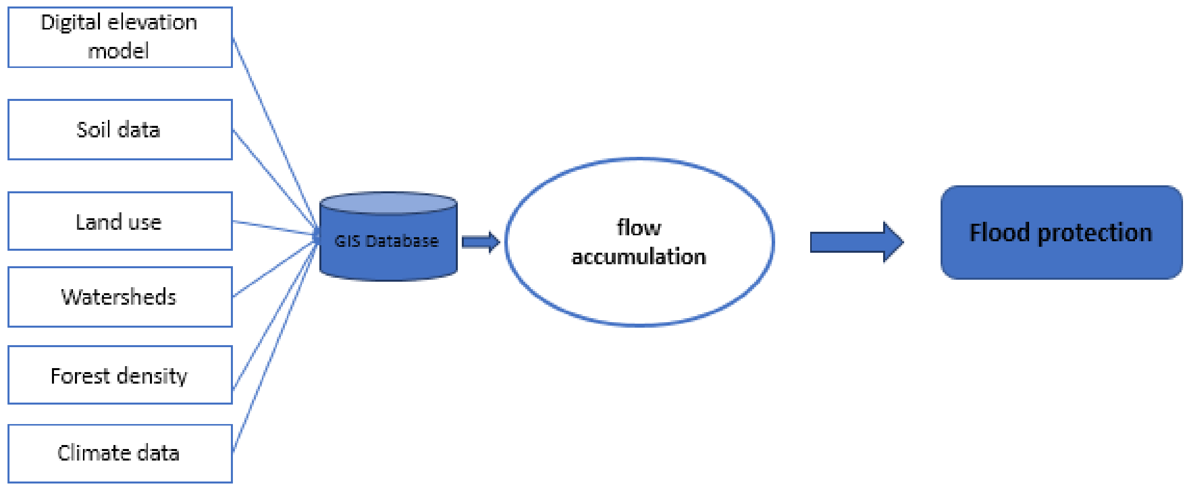

Figure 18 Regulating service – Flood protection

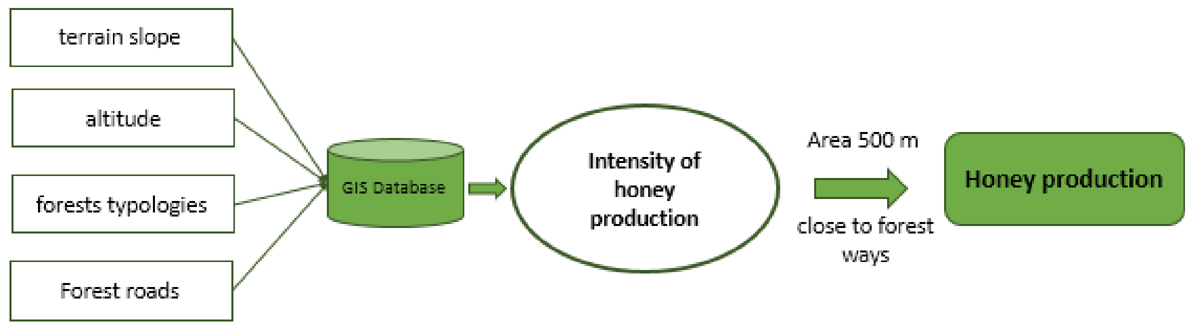

Figure 19 Provisioning service – Honey production

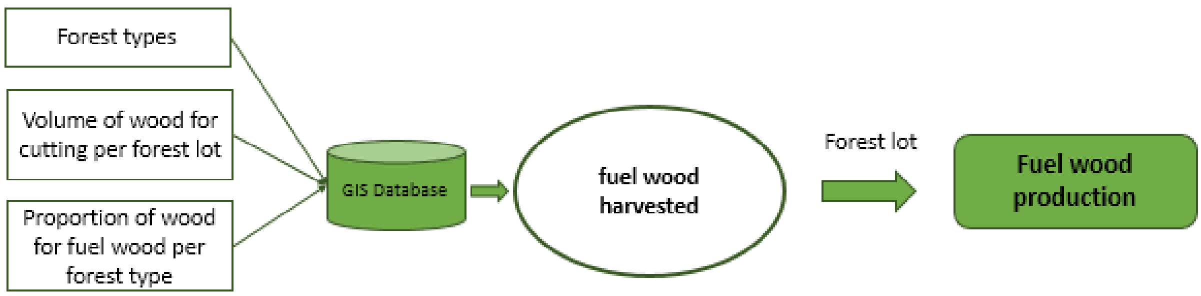

Figure 20 Provisioning service – fuel wood production

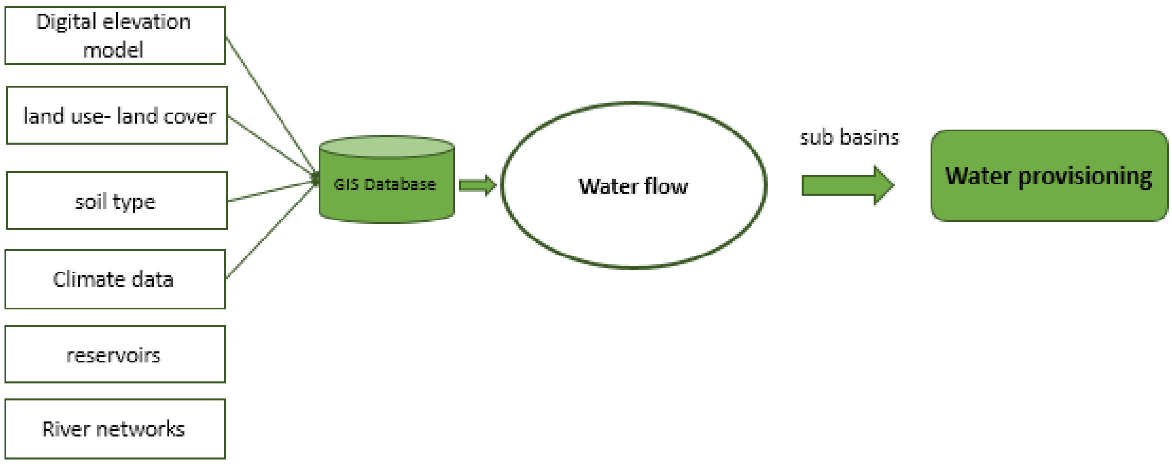

Figure 21 Provisioning service – water provisioning

5.2 Landscape metrics and biodiversity

A number of studies indicate that landscape metrics on an aggregated, overall landscape level are quite appropriate to describe the state of biodiversity (Walz 2011). Information on spatial distribution of sample plots is helpful to understand the spatial structure and patterns of forest habitat patches and can be of interest for biodiversity conservation (Corona et al. 2011).

Landscape metrics can be classified into (Corona et al. 2011)

- diversity and spatial pattern of land cover types by specific landscape metrics (diversity of land cover types, proportion, Shannon's diversity, dominance),
- total area and patchiness of habitat suitable for a particular species (patch density, mean patch size, largest patch index),
- spatial continuity and connectivity of important habitats (contagion).

Landscape metrics are commonly quantified on the basis of land cover thematic maps obtained from remote sensing or by multi- source techniques (Corona et al. 2011), moreover the use of Geographic Information Systems (GIS) is required to analyse landscape structure using landscape metrics (Walz 2011).

Biological diversity is always defined for a certain reference area, and landscape structure is a key element for the understanding of species diversity. Spatial heterogeneity, as an expression of landscape structure, indicates the variability of the system's properties in spatial terms (Walz 2011) and is essential for the explanation of the occurrence and distribution of species from the local to the global level (Ernoult et al., 2003, 240).

(Walz 2011) provides some examples from the literature of linkages landscape between structure and species diversity / patterns of species distribution

- Habitats with spatially heterogeneous abiotic conditions provide a greater variety of potentially suitable niches for plant species as habitats with homogenous characteristics. Variations in physical structure (e.g., slope direction, soil structure) have proven to be an appropriate factor for the prediction of the richness, diversity and dominance of plant species
- The composition and diversity of plant species depend on the area size GIS Indicators.
- The shape of habitats can affect the number of species. Therefore, shape complexity can be used to analyse land cover data as an index for species richness

- the distance to viable habitats (isolation) also determines the composition and abundance of plant species
- increase in area proportions and sizes of settlements and green spaces, traffic density and shrubs structures) correlates in particular with an increase in the number of species of neophytes
- the structure and variety of land use in floodplains and natural distribution mechanisms are linked to high biodiversity of indigenous species, but also promote the establishment and spread of neophytes and archaeophytes.

The above relationships (landscape-biodiversity) can be made comprehensible by means of landscape metrics. More than one hundred metrics have been developed for the purpose of describing processes and landscape functions in the form of mathematical terms (Burkhard and Maes 2017). Honnay et al., (2003) show that regional plant variety can be predicted satisfyingly on the basis of relatively simple landscape indices as those described above:

Patch diversity and distribution measures

- Shannon Diversity Index (SDI)

SDI refers to diversity of patches in the area. If SDI value is zero it is understood area consist of single patch. Distribution of patches can be identified in each other and field (Gokyer 2013)

$$SDI = -\sum_{i=1}^{m} P_i \; lnP_i$$

- Simpson diversity index

SIDI equals 1 minus the sum, across all patch types, of the proportional abundance of each patch type squared (McGarial and Marks 1995).

$$SIDI = 1 - \sum_{i=1}^{sm} P_i^2$$

- Shannon Evenness Index (SHEI)

SHEI identifies to distribution (regular or irregular) of patches in the area. If SEI value approaches 1 it is understood patches distribution are regular in the field (Gokyer 2013).

$$SHEI = \frac{-\sum_i^m P_i \; lnP_i}{lnm}$$

- Simpson evenness index (SIEI)

SIEI equals 1 minus the sum, across all patch types, of the proportional abundance of each patch type squared, divided by 1 minus the quantity 1 divided by the number of patch types. In other words, the observed Simpson's Diversity Index divided by the maximum Simpson's Diversity Index for that number of patch types (McGarial and Marks 1995).

$$SIEI = \frac{1 - \sum_i^m P_i^2}{1 - \frac{1}{m}}$$

- Patch richness (PR)

PR measures the number of patch types present. PR is a key element of landscape structure because the variety of landscape elements present in a landscape can have an important influence on several ecological processes. PR equals the number of different patch types present within the landscape boundary (McGarial and Marks 1995)

$$PR = m$$

- Largest patch index (LPI)

LPI equals the area (m^2) of the largest patch of the corresponding patch type divided by total landscape area (m^2), multiplied by 100 (to convert to a percentage); in other words, LPI equals the percentage of the landscape comprised by the largest patch (McGarial and Marks 1995).

$$LPI = \frac{\max_{j=1} a_{ij}}{A} 100$$

Patch Shape indices

- Mean shape index (MSI)

MSI equals the sum of the patch perimeter (m) divided by the square root of patch area (m^2) for each patch of the corresponding patch type, adjusted by a constant to adjust for a circular standard (vector) or square standard (raster), divided by the number of patches of the same type; in other words, MSI equals the average shape index (SHAPE) of patches of the corresponding patch type (McGarial and Marks 1995)..

Vector $MSI = \frac{\sum_{j=1}^{n} \frac{P_{ij}}{2\sqrt{\pi}\, a_{ij}}}{n_i}$ Raster $MSI = \frac{\sum_{j=1}^{n} \frac{0.25 P_{ij}}{\sqrt{a_{ij}}}}{n_i}$

- Area-weighted mean shape index

AWMSI equals the sum, across all patches of the corresponding patch type, of each patch perimeter (m) divided by the square root of patch area (m^2), adjusted by a constant to adjust for a circular standard (vector) or square standard (raster), multiplied by the patch area (m^2) divided by total class area (sum of patch area for each patch of the corresponding patch type). In other words, AWMSI equals the average shape index (SHAPE) of patches of the corresponding patch type, weighted by patch area so that larger patches weigh more than smaller patches (McGarial and Marks 1995).

Vector $AWMSI = \sum_{j-1}^{n}\left[\left(\frac{P_{ij}}{2\sqrt{\pi a_{ij}}}\right)\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right]$ Raster $AWMSI = \sum_{j-1}^{n}\left[\left(\frac{0.25 P_{ij}}{\sqrt{a_{ij}}}\right)\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right]$

- Mean patch fractal dimension (MPFD)

MPFD equals the sum of 2 times the logarithm of patch perimeter (m) divided by the logarithm of patch area (m2) for each patch of the corresponding patch type, divided by the number of patches of the same type; the raster formula is adjusted to correct for the bias in perimeter (McGarial and Marks 1995).

Vector $MPFD = \frac{\sum_{j=1}^{n}\left(\frac{2 \ln p_{ij}}{\ln a_{ij}}\right)}{n_i}$ Raster $MPFD = \frac{\sum_{j=1}^{n}\left(\frac{2 \ln 0.25 p_{ij}}{\ln a_{ij}}\right)}{n_i}$

- Area-weighted mean patch fractal dimension

AWMPFD equals the sum, across all patches of the corresponding patch type, of 2 times the logarithm of patch perimeter (m) divided by the logarithm of patch area (m^2), multiplied by the patch area (m^2) divided by total class area (sum of patch area for each patch of the corresponding patch type); the raster formula is adjusted to correct for the bias in perimeter In other words, AWMPFD equals the average patch fractal dimension (FRACT) of patches of the corresponding patch type, weighted by patch area so that larger patches weigh more than smaller patches (McGarial and Marks 1995).

Vector $AWMPFD = \sum_{j-1}^{n}\left[\left(\frac{2lnP_{ij}}{\ln a_{ij}}\right)\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right]$ Raster $AWMSI = \sum_{j-1}^{n}\left[\left(\frac{2ln(0.25P_{ij})}{\ln a_{ij}}\right)\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right]$

Fragmentation and Isolation indices

- Mean nearest neighbor distance (MNN)

MNN equals the sum of the distance (m) to the nearest neighboring patch of the same type, based on nearest edge-to-edge distance, for each patch of the corresponding patch type, divided by the number of patches of the same type (McGarial and Marks 1995)

$$MNN = \frac{\sum_{j=1}^{n'} h_{ij}}{n'_i}$$

- Mean proximity index (MPI)

MPI equals the sum of patch area (m^2) divided by the nearest edge-to- edge distance squared (m^2) between the patch and the focal patch of all patches of the corresponding patch type whose edges are within a specified distance (m) of the focal patch, summed across all patches of the same type and divided by the total number of patches in the class. In other words, MPI equals the average proximity index for patches in the class. When the search buffer extends beyond the landscape boundary for focal patches near the boundary, only patches contained within the landscape are considered in the computations (McGarial and Marks 1995)

$$MPI = \frac{\sum_{j=1}^{n}\sum_{s=1}^{n}\frac{a_{ijs}}{h_{ijs}^2}}{n_i}$$

- Interspersion and juxtaposition index (IJI)

IJI equals minus the sum of the length (m) of each unique edge type involving the corresponding patch type divided by the total length (m) of edge (m) involving the same type, multiplied by the logarithm of the same quantity, summed over each unique edge type; divided by the logarithm of the number of patch types minus 1; multiplied by 100 (to convert to a percentage). In other words, the observed interspersion over the maximum possible interspersion for the given number of patch types. IJI considers all patch types present on an image, including any present in the landscape border (McGarial and Marks, 1995)

$$\mathrm{IJI} = \frac{-\sum_{k=1}^{m'}\left[\frac{e_{ik}}{\sum_{k=1}^{m'} e_{ik}}\right]\ln\frac{e_{ik}}{\sum_{k=1}^{m'} e_{ik}}}{\ln(m'-1)}$$

- Mean Patch Size (MPS):

MPS can be used to evaluate fragmentation. If MPS value increases, it is understood fragmentation increases in the field. If MPS value decreases it is understood fragmentation decreases in the field.

$$MPS = \frac{A}{N}\left(\frac{1}{10{,}000}\right)$$

- Total edge (TE)

TE equals the sum of the lengths (m) of all edge segments in the landscape. If a landscape border is present, TE includes landscape boundary segments representing true edge only (that is, contrast weight > 0). If a landscape border is absent, TE includes a user-specified proportion of the landscape boundary. Regard- less of whether a landscape border is present or not, TE includes a user-specified proportion of background edge (McGarial and Marks, 1995).

$$TE = E$$

In (Walz 2011) study important landscape metrics which repeatedly mentioned, or stand out as particularly significant are presented in (table 4)

Table 4 Important landscape metrics in the field of biodiversity. Source (Walz,2011)

Function	Index
Prediction and assessment of biodiversity in landscape mosaics of the agricultural landscape	o habitat diversity (number of habitat types per unit area) o habitat heterogeneity (number of habitat patches, lengths of ecotones per landscape unit) o portions of natural, semi-natural and intensive land used
Prediction of biodiversity	o Surface area of semi-natural ecosystems o Patch distribution, o edge and patch density
Prediction of species diversity	o Patch Density PD, Largest Patch Index LPI, o Simpson's Diversity Index SIDI, Proximity PROXMN, o Patch Richness PR, Edge density ED, o Euclidean Nearest Neighbour ENNCV, o Circumscribing Circle: CIRCMN o Number of species, population sizes, number of viable populations and habitat area o Landscape diversity, intensity of agricultural use, frequency weighted absolute species rich- ness of vascular plants
Planning of biotope networks	o Proximity Index (allows assessment of individual patches depending on functional connection with surrounding habitats) o Density of landscape elements, indices of connectivity/ isolation
Assessment of protected areas, habitat requirements of species of the core areas and edges	o Total Core Area TCA, o Total Class Core Area TCCA, o Number of Core Areas NCA, o Core Area Index CAI, o Cority
Landscape fragmentation	o Effective mesh size o Area of unfragmented open spaces
Quantification of the floristic diversity (habitat function)	o Shannon Diversity SHDI, o Number of different classes and their distribution
Smallness, shape richness as well as structuredness of a landscape (natural spatial diversity)	o Edge density ED, o Density of patch boundaries or linear elements in a landscape Length of contour lines per area, elevation difference between highest and lowest point, river length and area of surface waters
Diversity of land use	o Diversity of main land use types, length of forest edges, field sizes
Floristic species richness (general)	o Distance (isolation) to usable habitat, largest patch index LPI, patch size coefficient of variation PSCV
Floristic species richness (in natural ecosystems)	o Topographic and edaphic variables, in particular slope direction and water balance Shape complexity of the habitats
Floristic species richness (in landscapes)	o Surface area of land use, Geometric landscape complexity, Number of Shape Characterizing Points NSCP Length of edges

Faunal species richness	o	Road density, forested area, distance to nearest built-up area, density of human settlements, degree of soil imperviousness

The practical use of landscape metrics to evaluate biodiversity focuses on such aspects as the optimum number of metrics, their appropriate informational connotation, including explicit assessment of analysed structural characteristics, and functional aspects in accordance with the landscape systematics applied (heterogeneity and taxonomy) (Malinowska and Szumacher 2013) Table 6 presents a list of studies where landscape metrics were used for biodiversity assessments

Table 5 Exemplary metrics. Source Malinowska and Szumacher, 2013

Authors	Landscape metrics
Gallardo et al. 2011	Abundance, Absolute Richness, Shannon Diversity, Rare ed Richness, Functional Diversity, Size Diversity, Average tax. distinctness
Kim, Pauleit 2007	Shannon Diversity Index (SHDI), Shape Index (SHAPE), Area Weighted Shape Index (AWSI), Nearest Neighbour Distance (NND), Proximity Index (PI), Mean Proximity Index (MPi), Mean Neighbour Patch Value Index ($MNPV_i$)
Bailey et al. 2007	Patch Density (PD), Largest Patch Index (LPI), Edge Density (ED), Proximity Index Distribution (PROX), Euclidean Nearest Neighbour Distribution (ENN), Perimeter Area Ratio Distribution (PARA), Shape Index Distribution (SHAPE), Related Circumscribing Circle Distribution (CIRC), Patch Richness (PR), Simpson's Diversity Index (SIDI)
Schindler, Poirazidis, Wrbka 2008	SIDI, CIRCLE_AM and IJI, ECON_MN, FRAC_MN, SHAPE_AM and AREA_CV
Rocchini et al. 2006	Number of Patches (NP), Area Weighted Mean Shape Index (AWMSI) the Maximum, Minimum, Mean and Standard Deviation of Patch Size (MaxPS, MinPS, MPS, PSSD, respectively). Isolation - Mean Proximity Index (MPI).
Yue et al. 2005	Shannon Diversity Index SHDI (for number of types above 100) Simpson's Diversity Index SIDI
Kumar et al. 2006	Mean Edge Contrast (MECI), Edge Density (ED), Mean Patch Size (MPS), Patch Size Coefcient of Variation (PSCV), Mean Fractal Dimension (MPFD), Mean Nearest Neighbour Distance (MNN), Mean Shape Index (MSI), Contagion (CONTAG), Cohesion, Interspersion/Juxtaposition Index (IJI), Patch Richness Density (PRD), Shannon's Diversity Index (SHDI), Simpson's Diversity Index (SIDI); Percent of Landscape (PLAND)
Uuemma et al. 2011	ED or SIDI, TECI or ECON_MN, SHAPE_MN, and PRD
Constible et al. 2006	PLAND, AWMSI, CWED and IJI
Bar Massada et al., 2009	Simpson Index of Diversity (SIDI), Edge Density (ED)
Onaindia et al. 2004	Shannon's diversity index, Simpson's diversity index, Simpson's evenness index
Bailey et al. 2007	PD, LPI, PR, ED, ENNCV, PROXMN, CIRCMN and SIDI.

6 ECONOMIC VALUATION

6.1 Economic valuation of ecosystem services

Assuming available biophysical data from physical accounts, valuation techniques can be employed to provide monetary ecosystem service accounts and monetary ecosystem asset accounts (Badura et al., 2017)

Assessment of economic values of forest ecosystem services is critical for (Baral et al., 2017):

- ✓ analyzing the feasibility of market-based management schemes for these services, such as payments for ecosystem services, voluntary carbon markets and biodiversity banks

✓ lowering the transaction costs of developing market-based schemes

It is important to understand that the fundamental goal of environmental economic valuation is not to put a price on an ecosystem or its components, but rather to express the relative importance of various ecosystem goods and services for populations. Consequently, assigning monetary value to them stems more from a need to establish indicators that can be used in decision-making processes rather than from a need to create a hierarchy of these goods and services (Bourlion et al., 2016).

There are two main groups of economic valuation methods (Figure 9):

1. Revealed preference methods (RP) are based on actual market behavior of users of ecosystem goods and services (Figure 22). RP estimate values based on the preferences of individuals, shown by their behaviour. Examples are the Travel Cost Method and Hedonic Pricing. The former can be used to estimate the value of a protected area through the amount of time and money people spend in order to visit it. The Hedonic Pricing Method uses the changes in the market value of a good that is directly related to the ecosystem services to be valued. For example differences in property prices can be used as indicators of the cultural ecosystem services provided by the landscape (Russi and Brink, 2014).
2. Stated preference methods (SP) are used when there is no market for the good or service linked to the environmental asset requiring valuation and can be applied to all types of ecosystem goods and services Methodologies based on SP such as Contingent Valuation (Figure 22)., are based on the preferences that are directly stated by people through surveys. They investigate people's willingness to pay (WTP) for improved environmental conditions or their willingness to accept (WTA) compensation for a reduction in environmental quality(Russi and Brink, 2014).

The benefit transfer method is an alternative to RP and SP methods uses the results of previous studies to run a new valuation on a similar object, but in a different place or time, in as much as the situation is similar or comparable (Brahic and Terreaux, 2011).

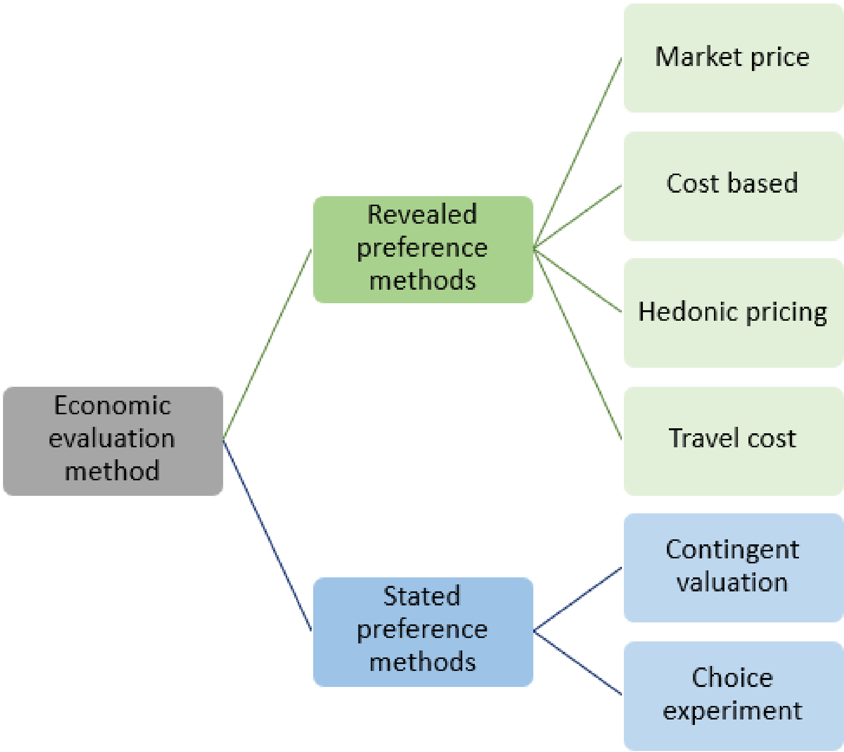

Figure 22 Economic valuation methods

More detailed information about methods for valuing ecosystem services is available in Deliverable D3.1.4 *Guidelines for sustainable capitalization of regulative services related to water resources management.*

Table 7 presents examples of flow accounts for a selection of ecosystem services, experimenting with valuation approaches. For these services, either data or models are available to produce at least annually aggregated values of physical flows.

Table 6 Ecosystem services with valuation approaches

Service	**Physical unit**
Provisioning services	
Timber	Timber growth and harvest (m^3 per ha)
no wood products (mushrooms, honey)	kgr per ha
Water	Water flow for drinking, irrigation and industrial uses from withdrawal points (m^3 per ha per year)
Livestock	Amount of animal (animal per unit)
Regulating services	
Water flow regulation	Storage capacity of surface water is related to the water volumes stocked in lakes, reservoirs (m^2)
Erosion control (soil protection)	Avoided erosion in m3/ha/year compared to bare soil
Carbon sequestration	C sequestration in ton/ha/year

Flood control	Land area protected
Cultural services	
Recreation	Number of visits in ecosystems (person-days) / ha, include budget for surveys in some countries
Tourism	Number of overnight stays generated per ha/year

6.2 Economic valuation of biodiversity

Economic valuation provides a monetary indicator of biodiversity values. The theoretical basis of economic valuation is monetary variation as a compensation or equivalent for a direct and indirect impact(s) on the welfare of humans due to a certain biodiversity change (Dasgupta, 2001). In economic terms, this can be considered as contributing to different elements of Total Economic Value (TEV), which comprises both use values (including direct use such as resource use, recreation, and indirect use from regulating services) and non-use values, e.g. the value people place on protecting nature for future use of provisioning, regulating and cultural service (option values) or for ethical reasons (bequest and existence values) (Figure 23). The economic importance of most of these values can be measured in monetary terms, with varying degrees of accuracy, using various techniques (including market pricing, shadow pricing and questionnaire based) (De Groot Rudolf et al., 2010).

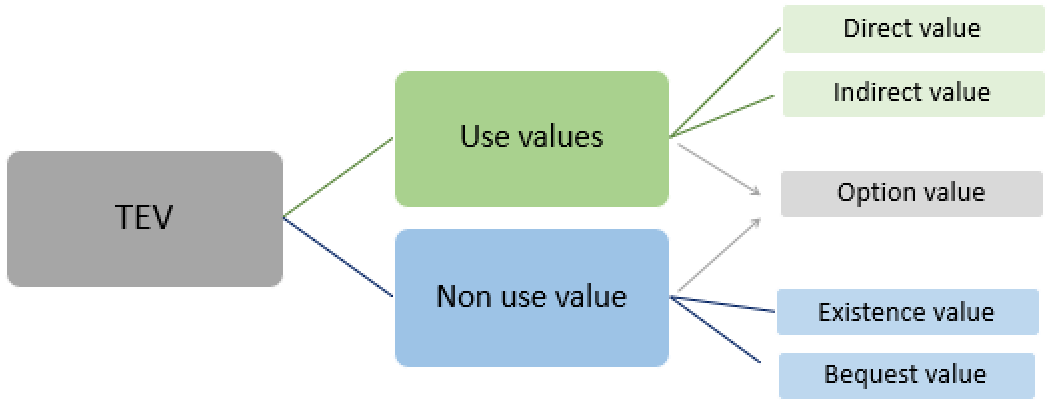

Figure 23 Components of the total economic value

The notion of direct value of biodiversity is sometimes used to refer to human uses of biodiversity in terms of production and consumption. The notion of indirect value of biodiversity has been associated with a minimum level of ecosystem infrastructure, without which there would not be the goods and services provided by it (Dasgupta, 2001)

TEV is the main framework for valuing biodiversity in monetary terms. Alternative value frameworks include social and ecological values (TEEB, 2010a). Social benefits include mental well-being, ethical, religious, spiritual and cultural values, which are often prominent in LDCs (UNEP, 1999); Biodiversity may also deliver ecological benefits which include the maintenance of many of the essential life support processes (e.g. soil formation, nutrient cycling)(Christie et al., 2012)

A range of techniques have been used to measure the economic, social and ecological benefits derived from biodiversity and associated ecosystem services. Monetary approaches may be used to capture the economic value of some or all of the elements of TEV.

Table 8 provides a summary of monetary valuation techniques and their suitability to measure different components of the TEV of biodiversity according the study of (Nijkamp et al., 2008). The table shows that certain valuation methods are more appropriate than others to address certain types of biodiversity value.

Table 7 Review of economic methods for biodiversity conservation. Source: Nijkamp et al., 2008

Method	Pros	Cons
Travel cost method	Use of real market data	Can estimate use values only May have substantial data requirements Requires estimates of value of travel/leisure time Cannot predict the changes in use values due to environmental changes without prior information Cannot predict the changes in use values due to environmental changes without prior information
Random utility model	Estimates recreational use value of (i) changing environmental quality of site attributes and (ii) site in total	Can estimate use values only May have substantial data requirements Requires estimates of value of travel/leisure time Problems arise with multi-purpose trips Cannot predict the changes in use values due to environmental changes without prior information Can be hard to handle participation decisions (i.e. whether to make the visit or not)
Hedonic pricing method	Use of real market data	Can estimate use values only Requires extensive house market data Cannot predict the changes in use values due to environmental changes without prior information Current evidence suggests it is not suitable for use in benefits transfer
Avertive expenditure method	Modest data requirements Use of real market data	Can estimate use values only problems arise when (i) individuals make multiple averting expenditures, (ii) there are secondary benefits of an averting expenditure and (iii) averting behaviour is not a continuous decision but a discrete one (e.g., double glazing is either purchased or not) Cannot predict the changes in use values due to environmental changes without prior information
Contingent valuation	Can estimate both use and non-use values Suitable for valuing environmental changes irrespective of whether or not they have a precedence Completed surveys give full profile of target population	Relatively expensive Complex and multi-dimensional scenarios may be too much of a cognitive burden for respondents The concept of diversity may similarly be difficult to put across to the respondents

Choice modelling	Can estimate both use and non-use values	Not yet as widely tested as CV Some techniques are not based on economic theory
	Suitable for valuing environmental changes irrespective of whether or not they have a precedence	The concept of 'diversity' may be difficult to put across to the respondents
	Completed surveys give full profile of target population	

Economic valuation of biodiversity is useful and perhaps indispensable for efficient forest management in that it makes possible, for example, comparisons of the costs and benefits between prior and modified situations (e.g. costs resulting from damage to biodiversity during forestry work) (Brahic and Terreaux, 2011). As (Dasgupta, 2001) stated, economic valuation of biodiversity is operationalized through explicit biodiversity changes, preferably marginal or small, and thus involving the design of alternative biodiversity policy management options, or scenarios.

Studies that value species or ecosystem diversity usually do not value the biodiversity itself, but rather focus on particular species and habitats. Several studies tackle the question of forest biodiversity valuation considering the value people assign to the preservation of genes, species or ecosystems (Garcia et al., 2011) more recent research effort focuses on the more holistic 'ecosystem services' approach to biodiversity valuation (Defra, 2007).

The economic benefits of forest biodiversity have been measured with stated preferences methods in several non-market valuation studies in European countries. A number of valuation studies assume that the biodiversity of a forest is linked with a set of environmentally-friendly management practices and attempt to value biodiversity preservation by explicitly stating to respondents that the implementation of a new conservation policy or type of forest management will result in a change of biodiversity in a particular area (Garcia et al., 2011).

6.3 Economic assessment tools

Assigning values to the benefits from ecosystem services often estimated by assigning monetary values to specific land covers using GIS (Nemec and Raudsepp-Hearne, 2013). Recently, studies have attempted to use GIS in conjunction with social science methodologies to assign non-monetary values (also static) to ecosystem service benefits. This has been accomplished through participatory mapping exercises (e.g. Bryan et al. 2011), where ecosystem service beneficiaries rate areas that are important or valuable to them in terms of ecosystem service provision and benefits (Nemec and Raudsepp-Hearne, 2013).

Several tools are applicable for assessing, sociocultural, economic and ecological values of forest ecosystem services.These tools have different strengths and are able to assess a particular value of ecosystem services. However these tools can complement each other and can be jointly implemented to assess multiple values of mountain forest ecosystem services (Baral et al., 2017). Tools for economic valuation except on the market analysis (Stated-preference, Revealed-preference technique and Benefit transfer) could also based on modeling analysis as it is

1. Toolkit for Ecosystem Service Site-based Assessment (TESSA)

The tool provides a guideline that illustrates eight steps for the successful assessment of ecosystem services at a site scale. It mainly focuses on stakeholder identification and engagement to explore various ecosystem services and to understand ecosystem services rights and value systems that different stakeholders obtain (Baral et al., 2017).

2. The Integrated Valuation of Ecosystem Services and Trade-offs Tool (InVEST)

InVEST is a family of tools developed in the Natural Capital Project to map and value the goods and services from nature which are essential for sustaining and fulfilling human life (www.naturalcapitalproject.org). InVEST models are based on production functions that define how an ecosystem's structure and function affect the flows and values of environmental services. Based on presentations about the tool box, InVEST seems to be primarily a model framework, which can be applied in specific circumstances or case studies based on stakeholder engagement and development of scenarios, which then feed into biophysical and economic models that are or have been adapted to local case study conditions (TEEB, 2010b).

3. Ecosystem Valuation Toolkit (EVT)

EVT provided by Earth Economics and comprises of a comprehensive, spatially-explicit, web-based repository of published and unpublished economic values for ESs.

4. The Multiscale Integrated Models of Ecosystem Services (MIMES)

MIMES integrate diverse types of knowledge and elucidate how benefits from ecosystem services are gained and lost (Boumans et al., 2015). The MIMES use input data from GIS sources and time series data to simulate ecosystem components (Figure 24) under different scenarios defined by stakeholder inputs (Baral et al., 2017).

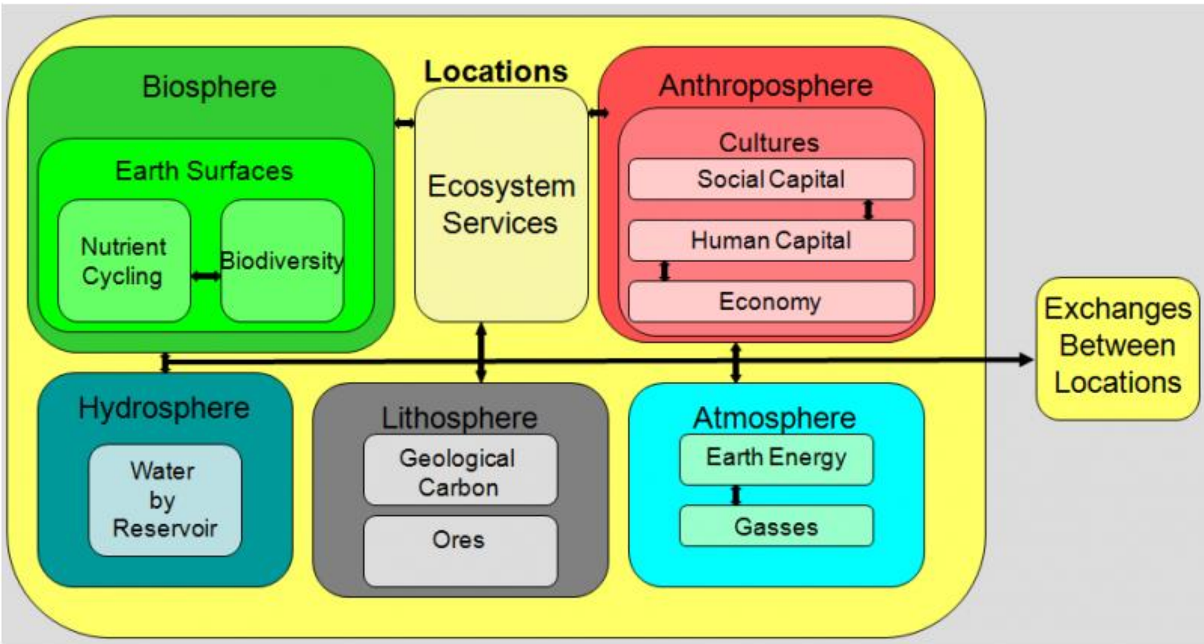

Figure 24.The MIMES organizational and interaction matrix. Source: Boumans et al., 2015

6.4 Economic models of biodiversity

(Eppink and van den Bergh, 2007) work refers to economic modelling and theoretical analysis of biodiversity loss and conservation. They reviewed studies within four economic model categories: cost-effectiveness (CE) models, renewable resource extraction (RR) models, macroeconomic growth (MG) models with natural resources and general equilibrium (GE) models with natural resource externalities

- Cost-effectiveness (CE) models characterised by some budget restriction. Many CE models use techniques from operations research to look for a cost-effective spatial distribution of habitat patches. Additionally, some models cover other issues such as prioritising species and policy instruments for conservation. Although many of these models do not explicitly define the decision-makerare regarded as a tool for conservation management. CE models can be applied at large spatial scales to study conservation of genetic, species, ecosystem as well as functional biodiversity (Eppink and van den Bergh, 2007).
- Renewable resource extraction (RR) models reflect that a species' population may have economic value, particularly when harvested. RR models can be used to study two types of problem. One is that of an individual resource owner who determines his private optimal rate of harvesting, whereas the other is that of a government planner who wants to achieve socially optimal harvesting (Eppink and van den Bergh, 2007).
- Macroeconomic growth (MG) models are used to analyse optimal growth paths of an economy over time. When extended with environmental effects of growth, MG models can indicate whether or not this growth is sustainable. Since economic growth is a macroeconomic variable, the decision-maker who desires optimality or sustainability is a government planner. This is a broad generalisation, since even our selective survey comprises models that use various assumptions and model specifications. Older MG models tend to focus on non-renewable resources and irreversibility, whereas newer models mainly consider renewable resources(Eppink and van den Bergh, 2007).
- General equilibrium (GE) models are systems of input–output relations, or markets, where prices and volumes of traded goods are determined simultaneously. If a traded good requires inputs for which there are no markets, such as many natural resources, then the use of that traded good is inefficient. GE models can be used to study such inefficiencies as well as policy instruments needed to remedy them. GE models describe the behaviour of an individual, but it is assumed that this individual is representative of a large group of individuals. Therefore, GE models essentially describe complete economic systems and thus the setting of a large spatial scale is appropriate for models of this type.

6.5 Case studies

In a study by van Beukering et al. (2003) the value of biodiversity in a national park in Indonesia was measured based on the amount of research funding spent on research in the park.

Xie et al. (2010) applies a method for biodiversity based on the number of plant and animal species preserved. They find a value of 521 USD per hectare

Though not directly a measure of biodiversity, Amirnejad et al. (2006) estimated the existence value of the forests of Iran through a contingent valuation survey, which resulted in a value of 240 USD per hectare.

In Pechanec et al., (2017) study the concept of payments for ecosystem services (PES) are applied for the monetary valuation of natural forest habitats, which were mapped in the Czech Republic in order to create the Natura 2000 European network. The method is based on expert evaluation of every type of mapped habitat by a point value for specific ecological evaluating criteria. The monetary value of every point of specific natural forest habitats was defined from the economic analysis of financial expenses of realised ecological restoration projects in the Czech Republic

Garcia et al., (2011) study raises the issues of valuing biodiversity through a national survey to improve the national accounts in France. The willingness to pay (WTP) for forest biodiversity is estimated by a Dichotomous Choice Contingent Valuation (DC-CV) method. In the same survey, the households were asked whether they have recreational activities in forests or not. The recreation answer is not independent from the DC biodiversity question and can affect the estimated biodiversity WTP of households.

(Biénabe and Hearne, (2006) use choice experiments to investigate the preferences and the willingness to pay of foreign tourists and Costa Ricans for increased support for biodiversity conservation and scenic beauty through a system of Payments for Environmental Services (PESs). In order to assess preferences for these different public goods services of private forests, survey respondents were asked to choose between spatially differentiated areas to receive PESs. Through different experts and focus groups, the establishment of PESs in remote areas was acknowledged to favor nature conservation and their establishment in accessible areas, to favor scenic beauty.

Cerda et al., (2013) conducted a choice experiment for the economic valuation of benefits of components of biodiversity that are provided by the natural systems protected in the Pe˜nuelas Lake National Reserve, located in the Mediterranean zone of Chile.

Cerda et al. (2014) performed a choice experiment to estimate the willingness to pay (WTP) of the residents of an ecosystems of Navarino island in southern Chile for the preservation of the island's biodiversity

Brahic and Terreaux, (2011)design a choice experiment to investigate these questions in a setting where respondents are asked to evaluate the preservation of the Danish heath and its endangered species They also conclude that using 'iconised' species for valuing biodiversity at habitat level may lead to very high, potentially overestimated.

Turpie, (2003) study investigates the public interest, experience and knowledge of biodiversity and uses contingent valuation methods to estimate its existence value, with emphasis on the internationally significant fynbos biome in the Western Cape of South Africa.

Czajkowski et al., (2009) value changes in a number of attributes which describe complex characteristics of biodiversity, based on ecological knowledge. The attributes used include structural, species and functional diversity. The empirical application is a choice experiment study- willingness to pay conducted in the Białowieża Forest, Poland.

7 DATA OVER THE STUDY AREAS

Socioeconomic data over the designated Natura 2000 areas were gathered and analyzed (Table 9). These include among others, census data, data on distribution of vegetation and data and agricultural activities etc. The following maps present part of the data, analyzed for each Natura2000 site.

Table 8 Designated Natura 2000 areas were socioeconomic data was gathered.

NAME	AREA CHARACTERIZATION	AREA CODE	AREA (ha)
Vasilitsa Mountain	SPZ	GR 1310001	8012.78
Valia Calda and Aoos artificial lake	ZSP	GR 1310002	14660.48
Natural Park of Pindos (Valia Calda) (Broader Area)	SPZ	GR 1310003	6838.25
Orliakas and Tsourgiakas Mountains	ZSP	GR 1310004	10230.24
National Forests of Vikos-Aoos	SPZ	GR 2130001	12794.25
Peaks of Smolikas Mountain	SPZ–ZSP	GR 2130002	19975.72
Central Part of Zagori	SPZ	GR 2130004	33114.95
Metsovo Area (Anilio-Katara)	SPZ	GR 2130006	7328.82
Mitsikeli Mountain	SPZ	GR 2130008	8435.99
Tymfi (Gkamila) Mountain	ZSP	GR 2130009	27416.44
Central Part of Zagori and East Part of Mitsikeli Mountain	ZSP	GR 2130011	53407.84

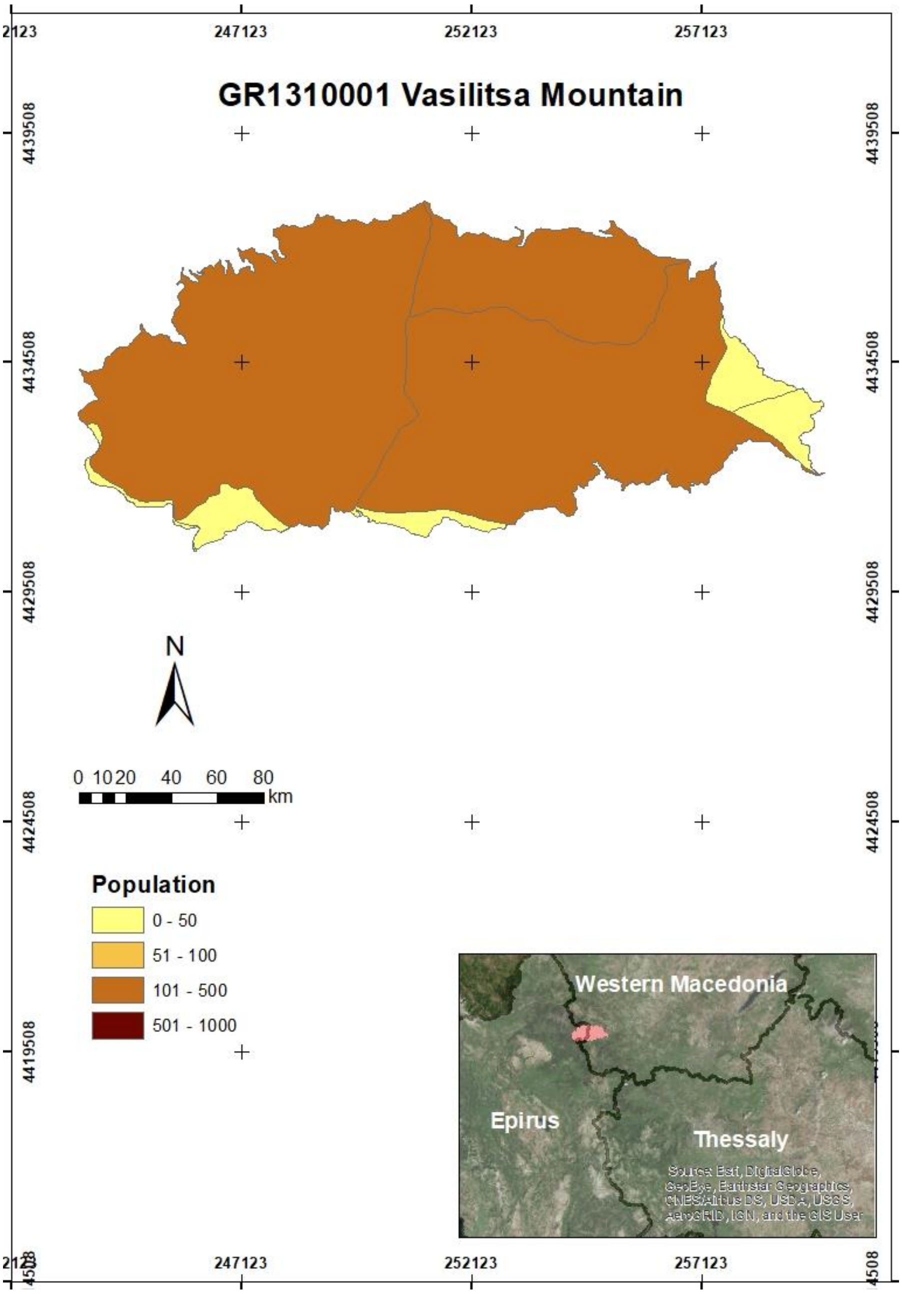

Figure 25 Census data over GR1310001 (2011)

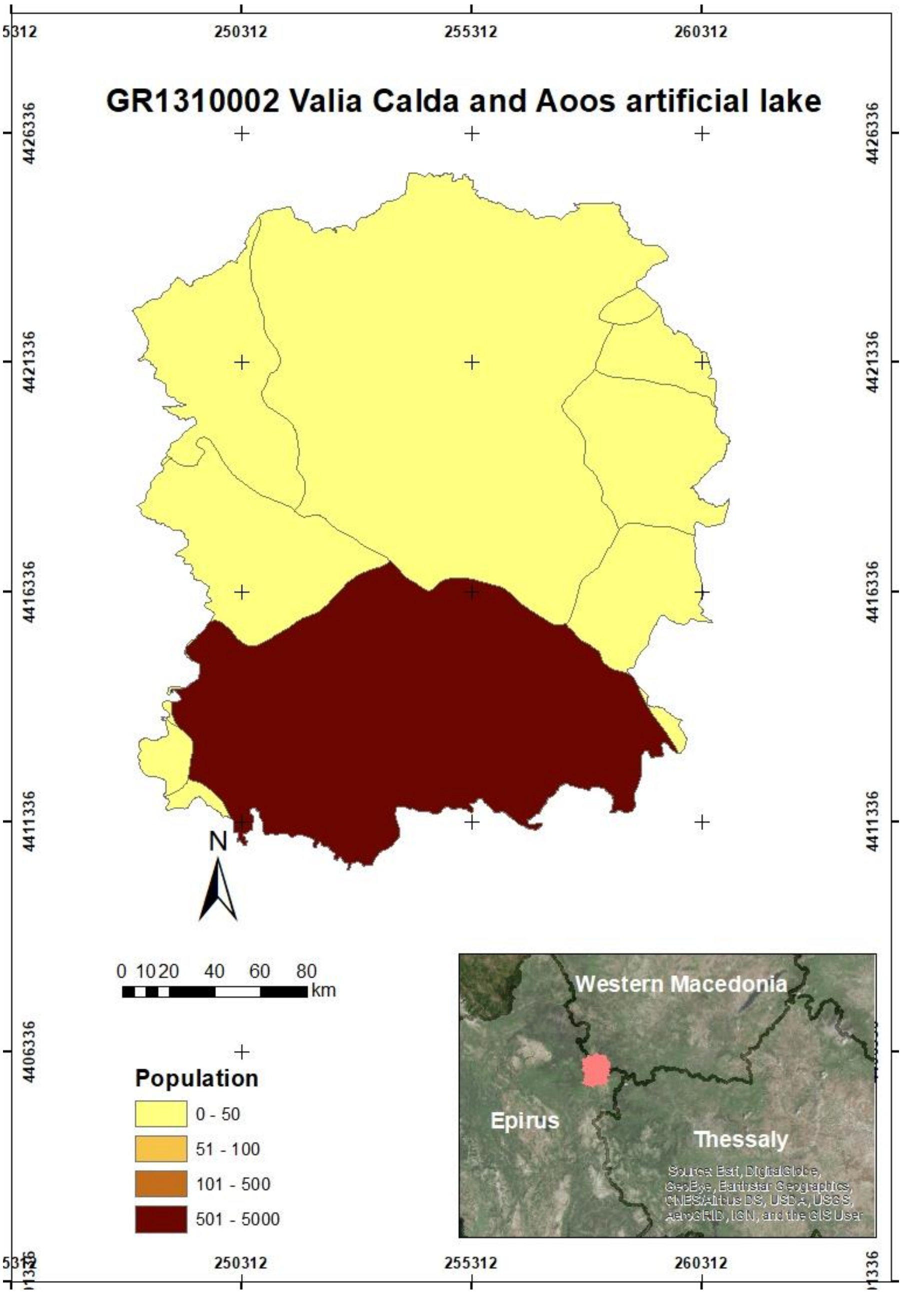

Figure 26 Census data over GR1310002 (2011)

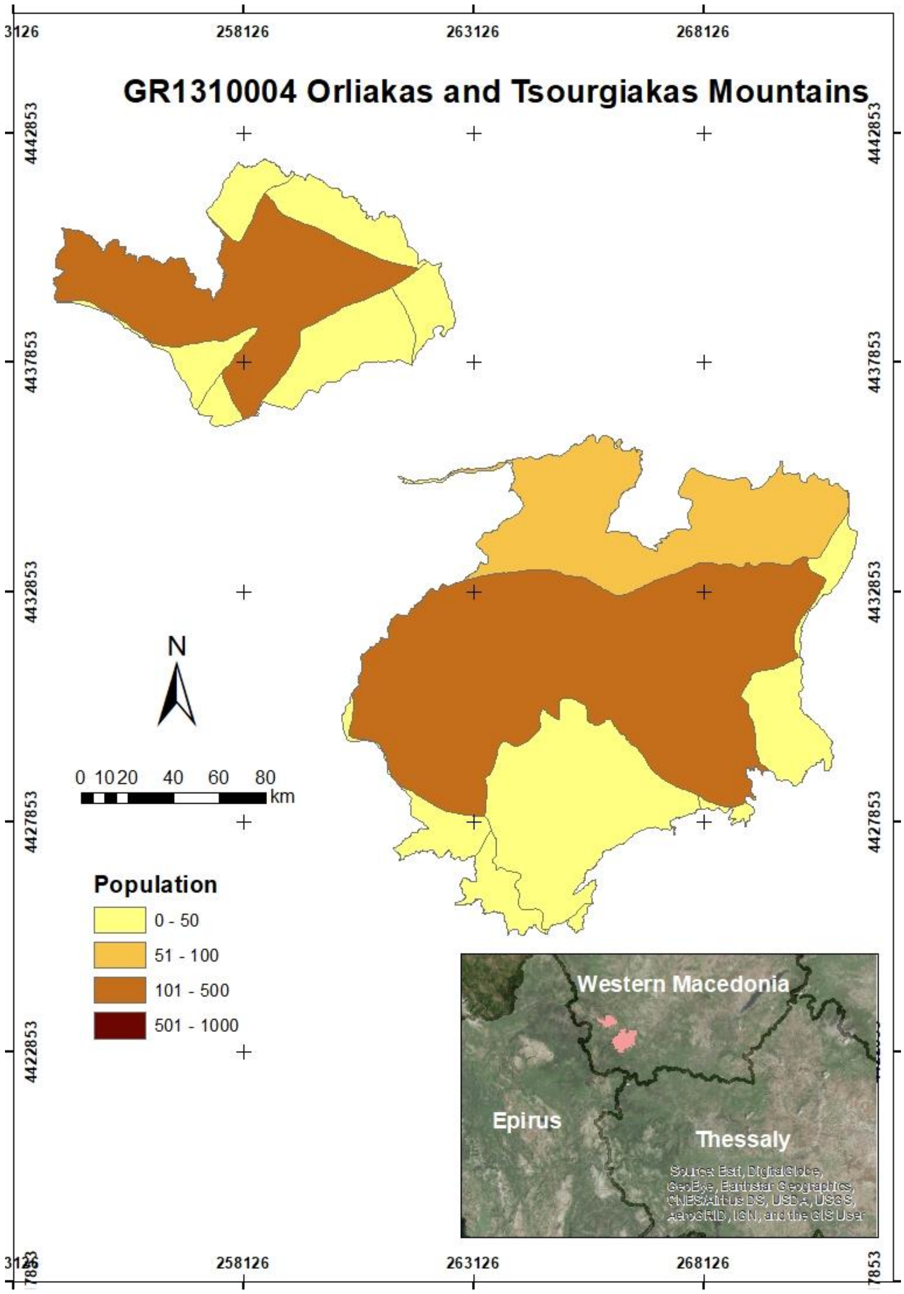

Figure 27 Census data over GR1310004 (2011)

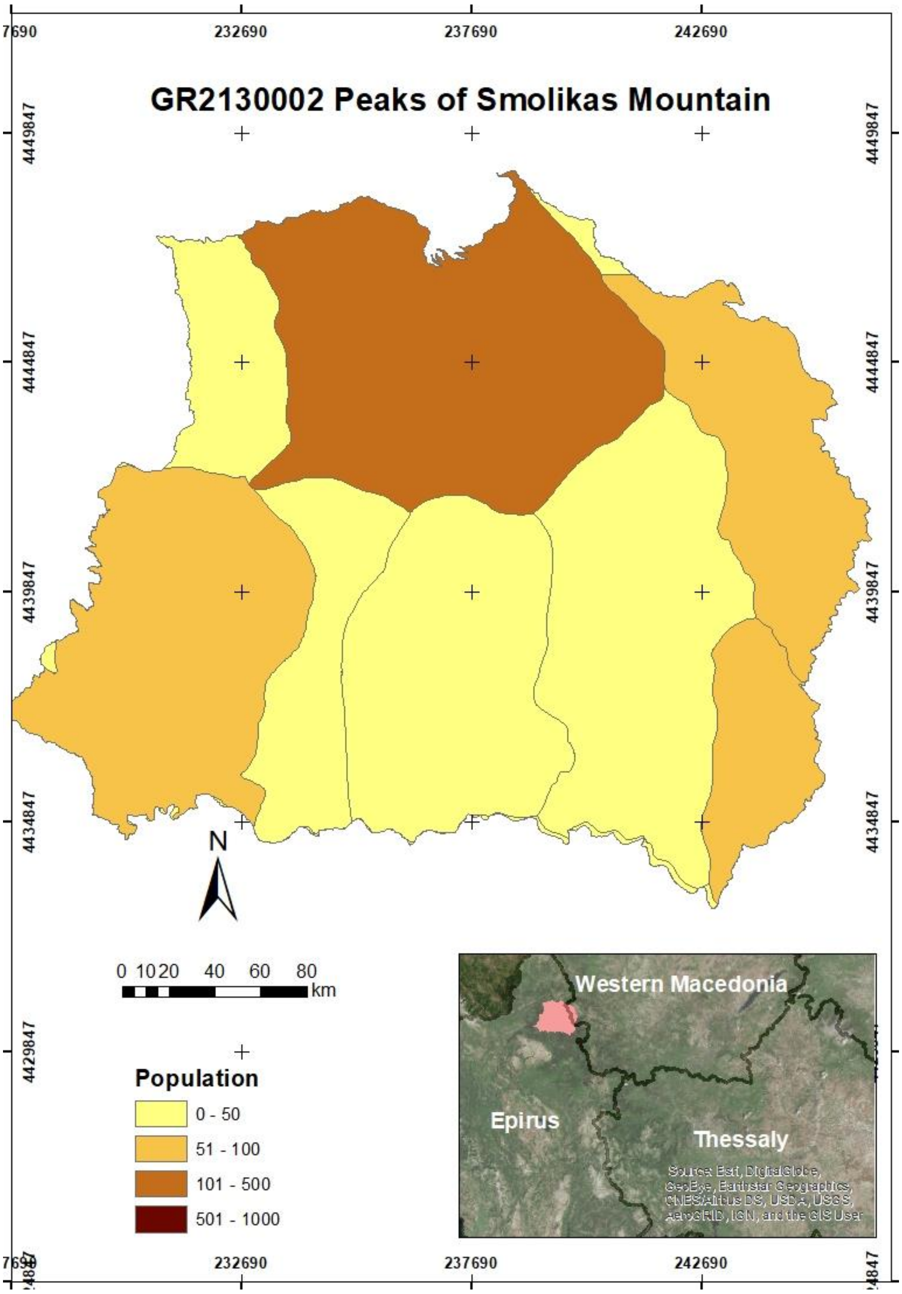

Figure 28 Census data over GR2130002 (2011)

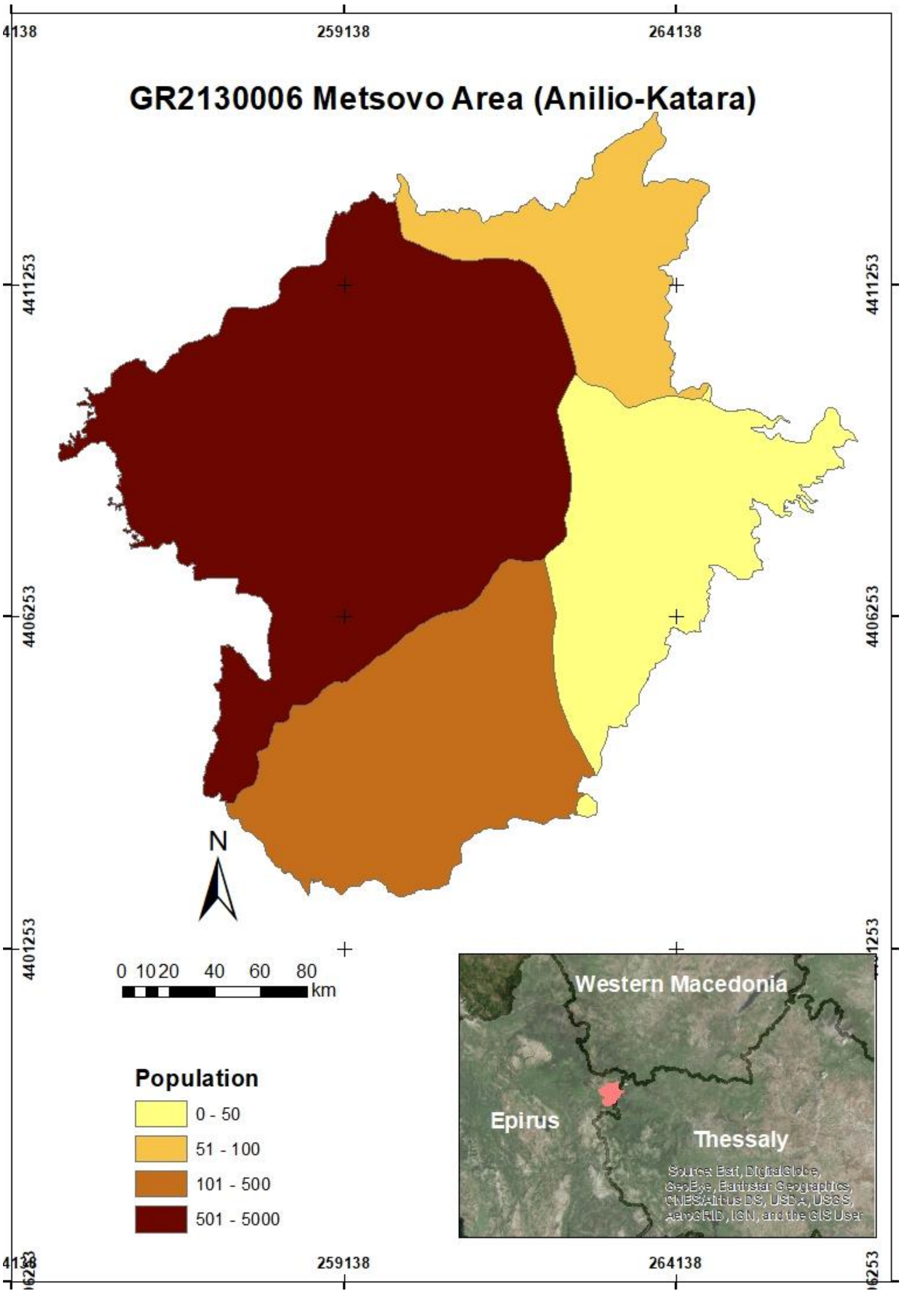

Figure 29 Census data over GR2130006 (2011)

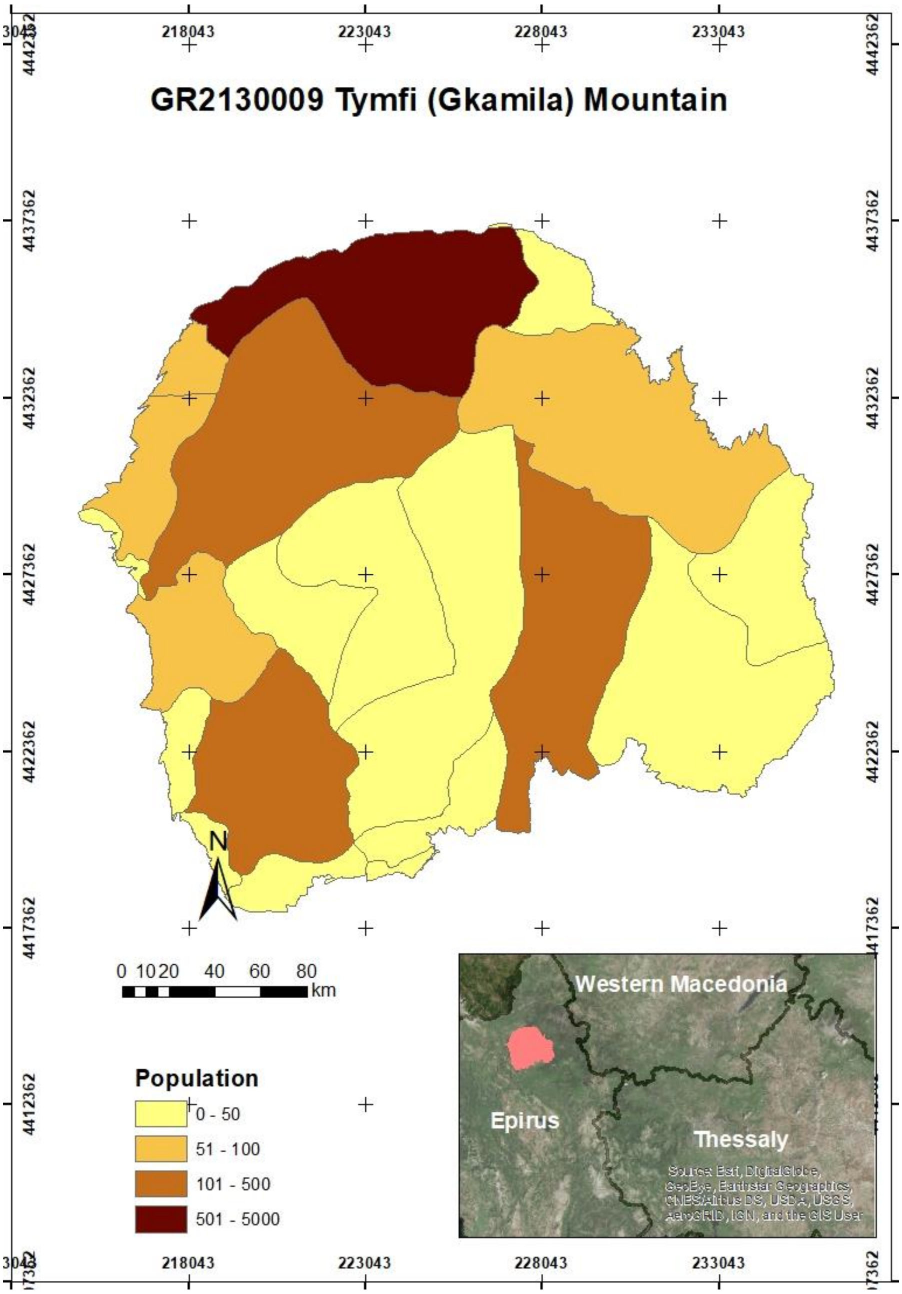

Figure 30 Census data over GR2130009 (2011)

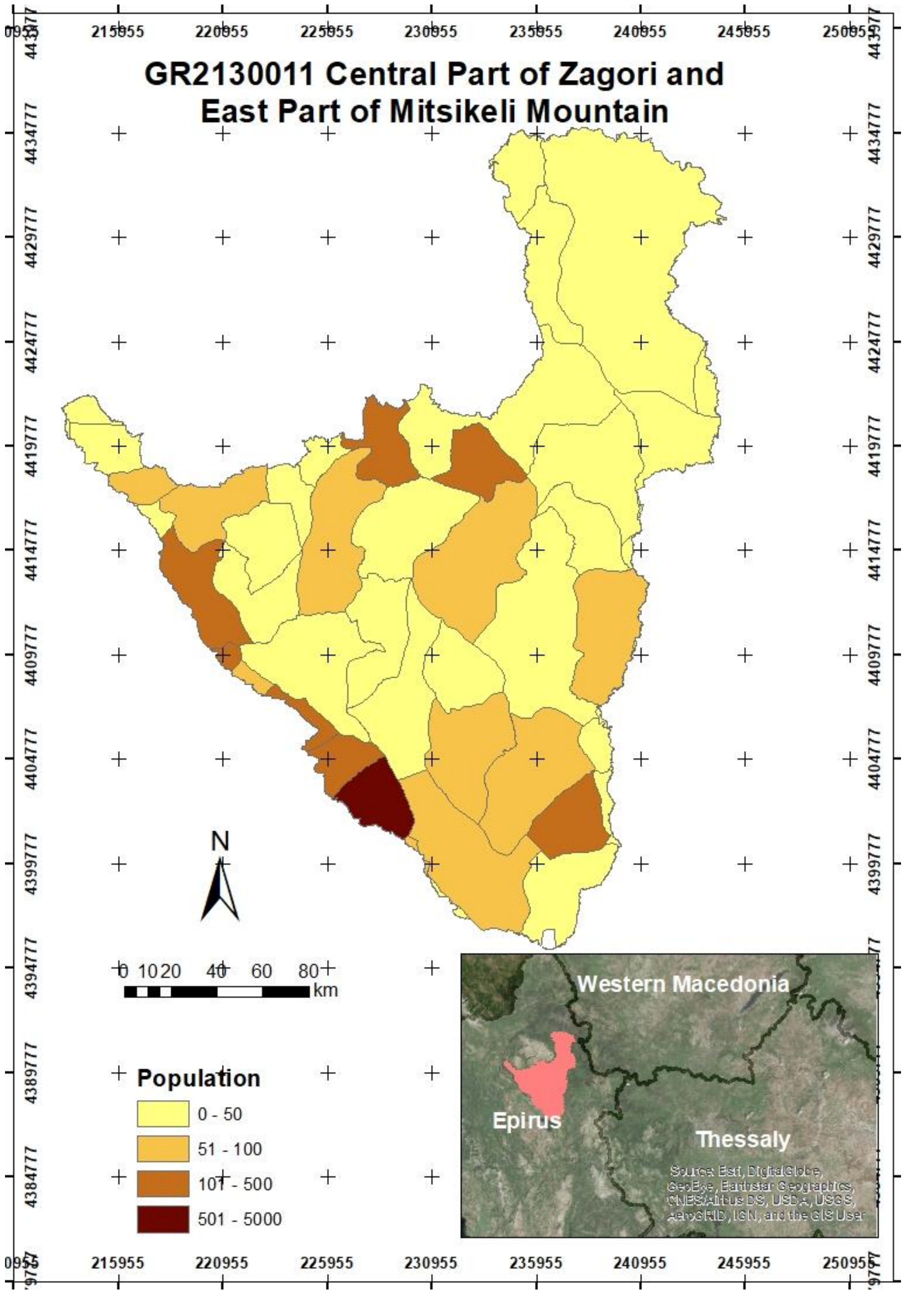

Figure 31 Census data over GR2130011 (2011)

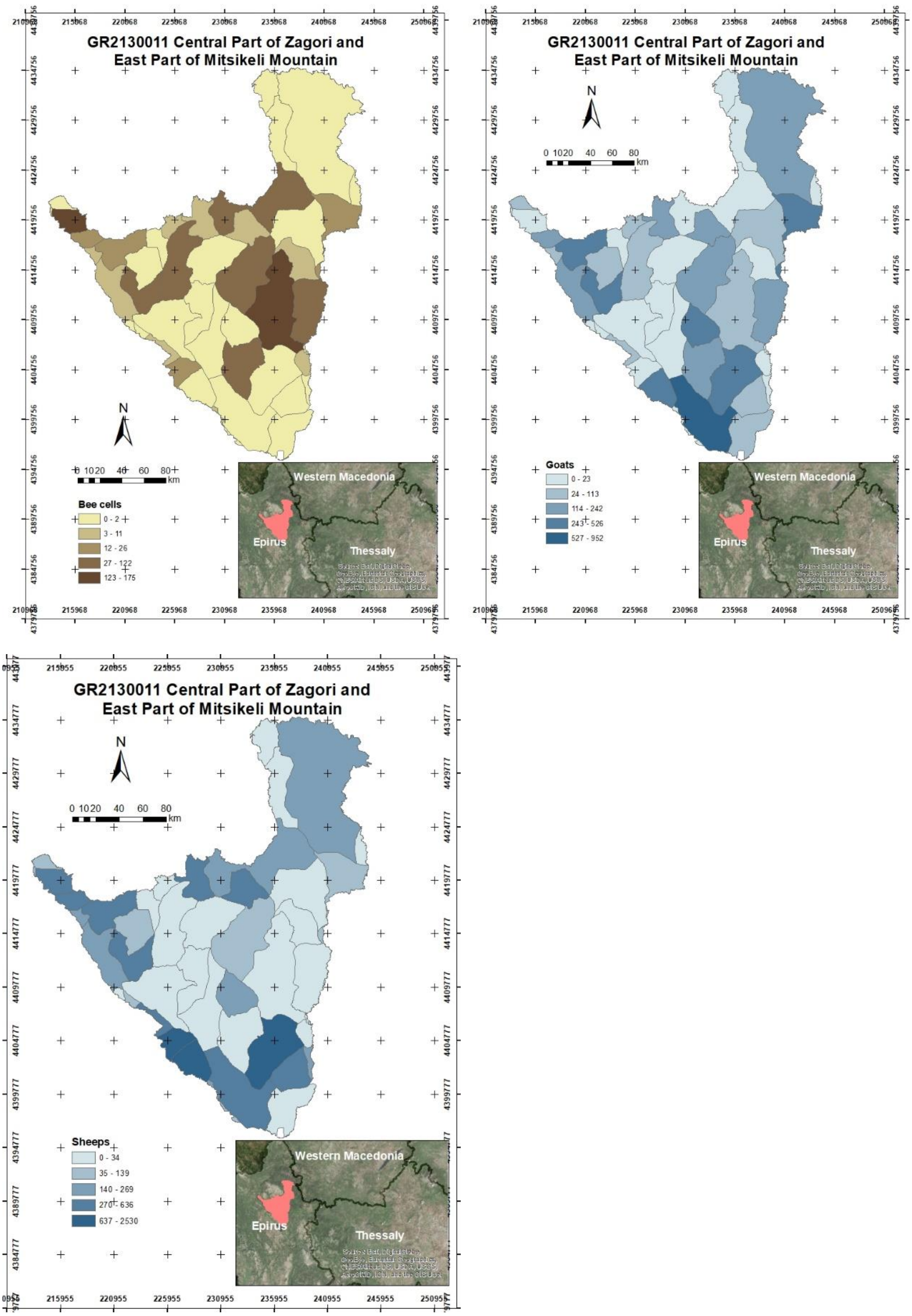

Figure 32 Economic activities data over GR2130011 (2000)

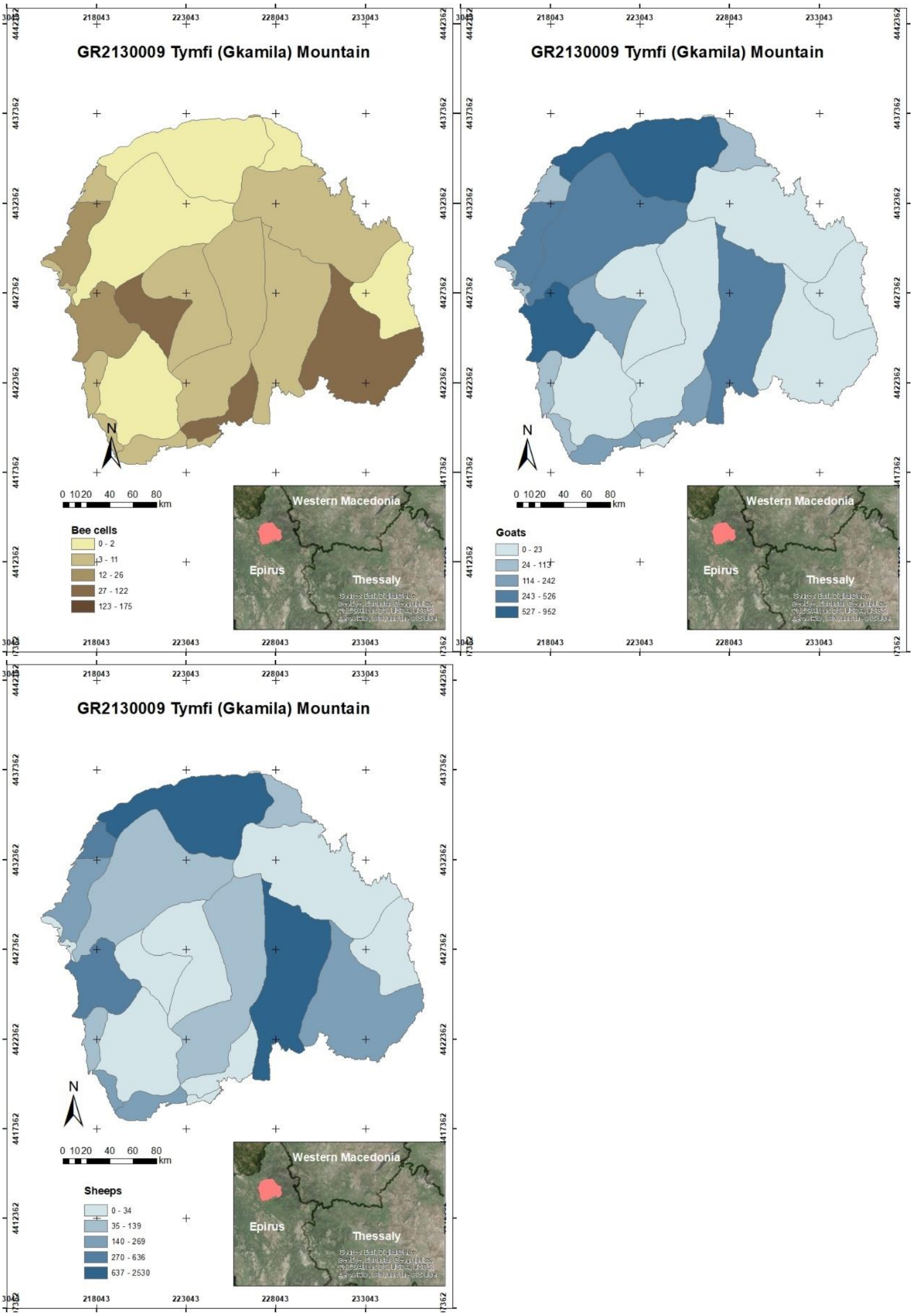

Figure 33 Economic activities data over GR2130009 (2000)

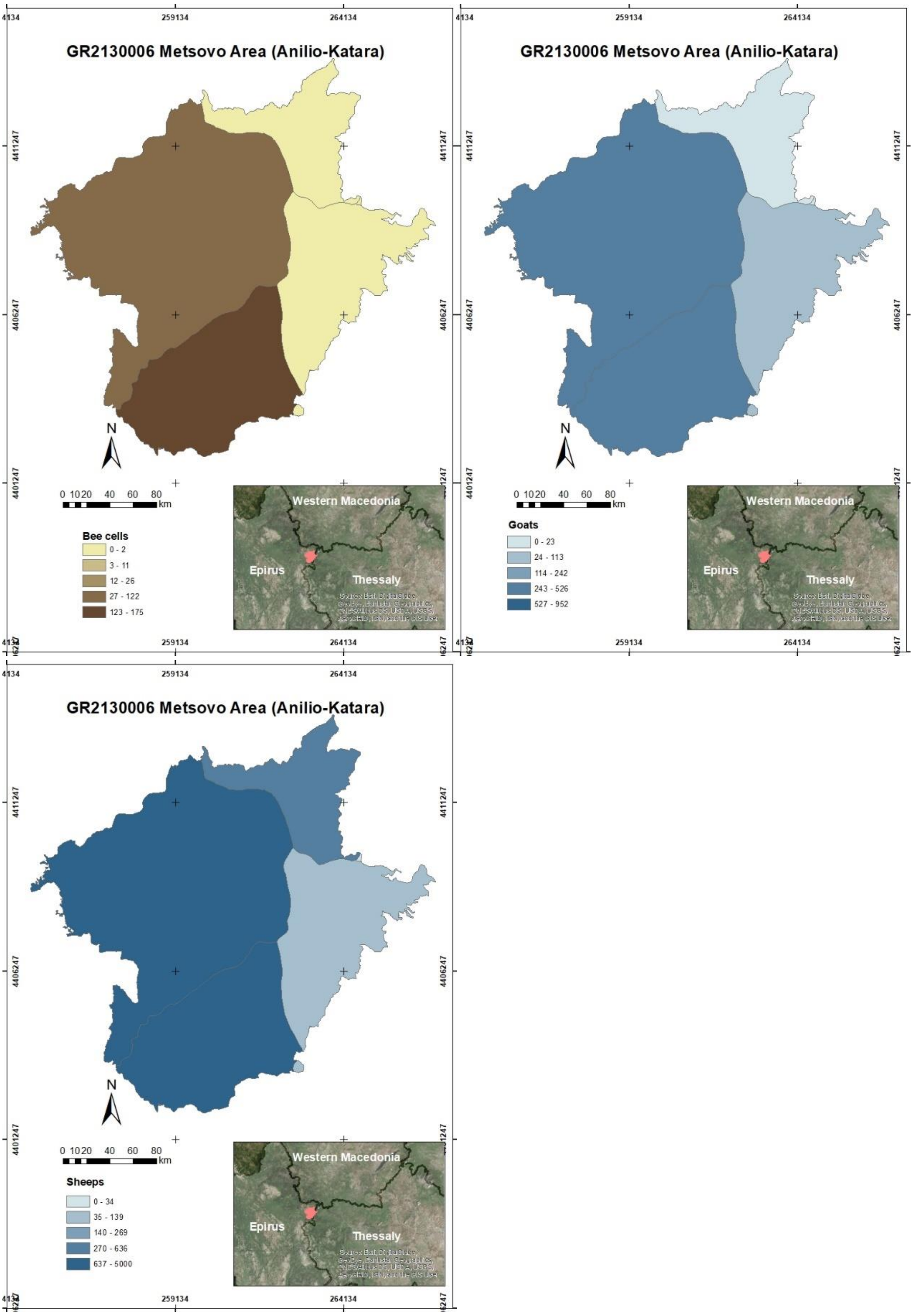

Figure 34 Economic activities data over GR2130006 (2000)

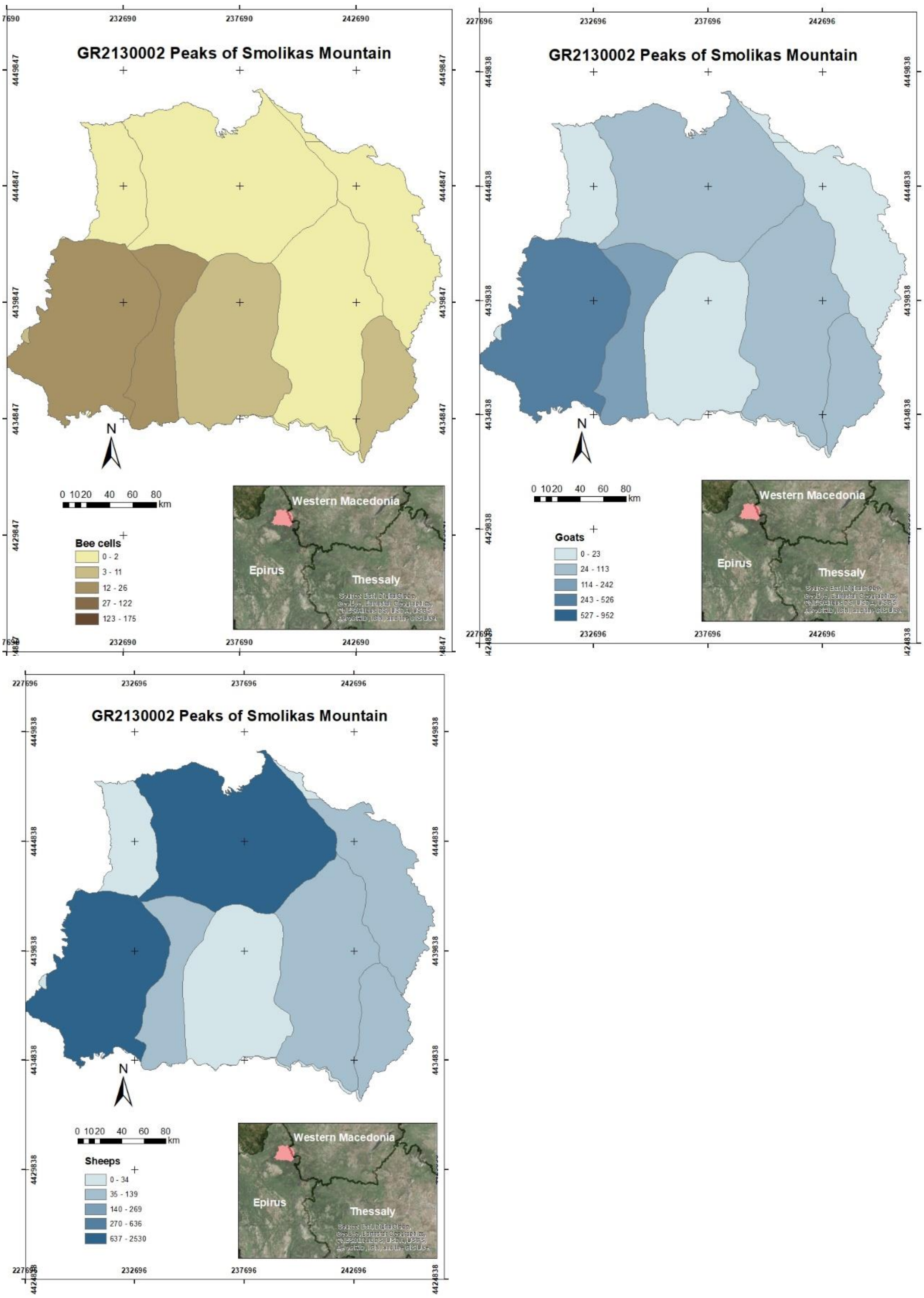

Figure 35 Economic activities data over GR2130002 (2000)

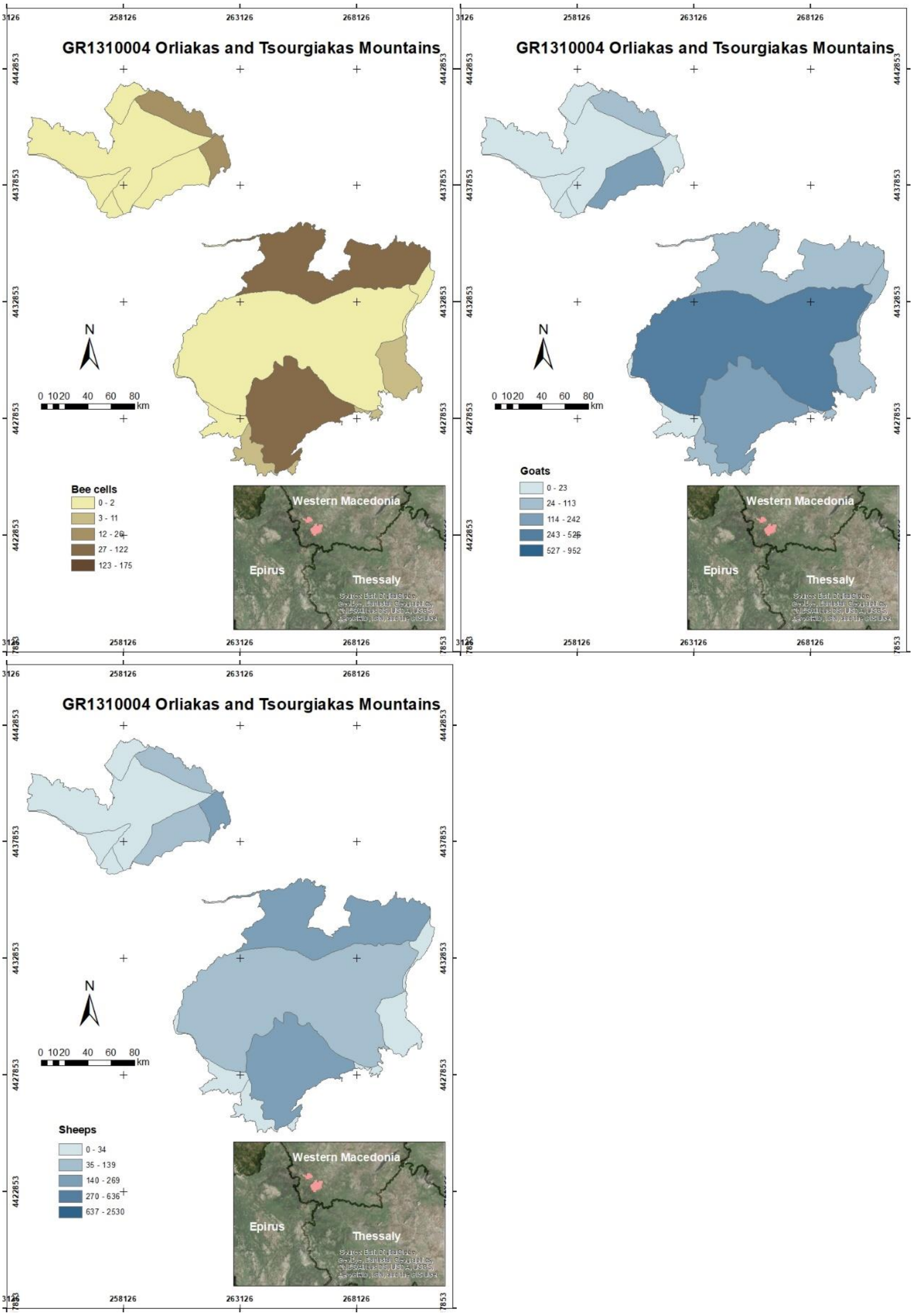

Figure 36 Economic activities data over GR1310004 (2000)

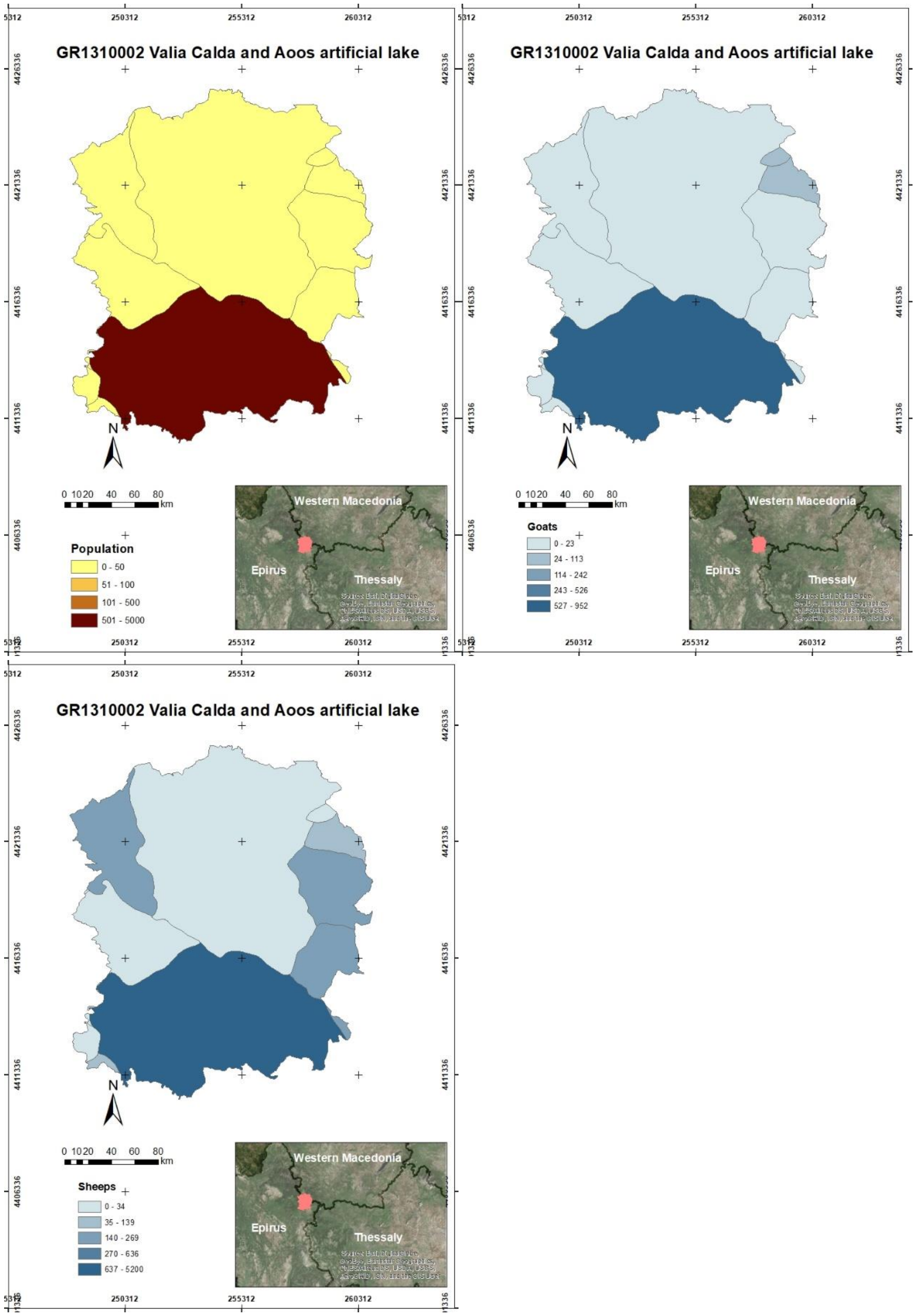

Figure 37 Economic activities data over GR1310002 (2000)

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