Recovery of degraded and transformed ecosystems in coal mining-affected areas

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Deliverable 4.2

Feasible valuation techniques

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Executive summary

In this deliverable, a new valuation methodology for non-provisioning ecosystem services is developed, as evidence was found regarding the lack of homogeneity caused by the existing approaches. Using different valuation methods for non-provisioning ecosystem services generates a lack of uniformity in the valuation process that Multi-criteria decision analysis can hardly correct.

In the first place, non-provisioning ecosystem services will be quantified before their monetisation using tables of coefficients for each land cover type derived from field experiments. Then, they will be transformed into a common metric, an index between one and ten, through local scaling. Local scaling sets upper and lower bounds using locally measured performance values instead of global scales that may cause irrelevance of differences between local measures.

In the second place, to monetise non-provisioning ecosystem services, the implementation of techniques based on the propagation of imprecise preference statements in hierarchical weighting will be used. Once a reference attribute has been selected, the remaining attributes are compared to the reference, considering the specific environment and the local scale used.

Finally, and to achieve consistency, monetisation of all non-provisioning ecosystem services will build on the above comparison and the monetary valuation of the attribute with the most direct and market-related valuation possible: carbon sequestration, which was valuated using the EU Emissions Trading System.

1 Introduction

Work package 4 aims to develop the formulation that will be used later for the cost-benefit assessment.

Within this work package, Task 4.2 foresees to define the best feasible valuation technique for every suitable indicator.

Since people are already familiar with money as a unit of account in our societies, expressing relative preferences regarding money values will give helpful information to RECOVERY's purposes.

The definition of the best feasible valuation technique for every suitable indicator was going to be addressed initially, taking into consideration the Total Economic Value (TEV) concept, following the TEEB (2010) taxonomy.

The concept of TEV of ecosystems is defined as the sum of values of service flows that natural capital generates both now and in the future, appropriately discounted. TEV encompasses all components of (dis)utility derived from ecosystem services using a standard unit of account: money or any market-based unit of measurement that allows comparisons of the benefits of various goods.

Within the TEV framework, values are derived, if available, from the information of individual behaviour provided by market transactions relating directly to the ecosystem service. In the absence of such information, price information must be derived from parallel market transactions associated indirectly with the good to be valued or by simulating a market and demand when no surrogate markets exist.

These situations correspond to a categorisation of the available approaches, among which the best feasible valuation technique will be defined for every suitable indicator:

1. Direct market valuation approaches: the main advantage of using these approaches is that they use data from actual markets and thus reflect basic preferences or costs to individuals.

2. Revealed preference approaches: are based on observing individual choices in existing markets related to the ecosystem service subject to valuation. In this case, economic agents "reveal" their preferences through their options.

3. Stated preferences approach: they simulate a market and demand for ecosystem services by employing surveys on hypothetical changes in the provision of ecosystem services. They can be used when no surrogate markets exist from which the value of ecosystems can be deduced.

Nevertheless, after a careful literature revision, evidence was found about the need to develop a new methodology to address appropriate valuations of ecosystem services.

The explanation of this methodology is described in this deliverable.

2 Assessing the value flows of ecosystem services

Attempts to assess the value flows of ecosystem services have been ongoing.

De Groot et al. (2002) presented a typology for valuing the goods and services of ecosystem functions based on their ecological, socio-cultural and economic value. Following this line of work, the Economics of Ecosystems and Biodiversity Ecological and Economic Foundations (TEEB, 2010) provided a basis for the monetary valuation of ecosystems and biodiversity by assessing their total economic value (TEV).

TEV is “the sum of values of all flows of services generated by natural capital, both now and in the future, discounted appropriately”. Valuation methods were classified according to three approaches: market valuation (based on price, cost and output), revealed preference (travel cost and hedonic pricing) and stated preference (contingent valuation, choice modelling, contingent ranking and deliberative group valuation).

The economic valuation methods considered were direct and indirect market valuation, contingent valuation and group valuation. Their aggregation and weighting to obtain the total value were highlighted as an essential issue due to the “weak comparability” of values. Multi-criteria decision analysis (MCDA) was pointed out as a tool that allows multiple values to be integrated after assigning each of them a relative weight. In addition, transparent deliberative processes will, in their view, facilitate the reduction of risk related to the inherent weaknesses of the MCDA.

However, the MCDA was developed to determine the best choice based on the scores of the different criteria and the relative weights given to those criteria. It is complicated to assign relative weights to other criteria that have been evaluated with varying assessment methods.

Many authors have addressed associating or comparing values attributed to ecosystem services. Hein et al. (2006) discussed the spatial scales at which ecosystem services are provided and the implications for the different stakeholder values attributed to ecosystem services. According to them, if all values are expressed by comparable monetary indicators (e.g. consumer or producer surplus), they can be summed. If not, they can be compared using the MCDA.

Sijtsma et al. (2013), seeking to better support decision-making on ecosystem services, argued that a careful combination of MCDA and cost-benefit analysis (CBA) facilitates evaluations of projects involving natural ecosystem services and agriculture changes. Nevertheless, their methodology avoids monetisation, using cardinal/ratio measures.

Wam et al. (2016) advocated exploring the valuation of trade-offs without direct pricing or MCDA but within a scheme of monetary exchange protocols. Saarikoski et al. (2016) argued that MCDA better values ecosystem services than CBA and linked monetary valuation techniques. They concluded that there is a need for research on hybrid methodologies combining MCDA and monetary valuation. Other authors, such as Spangenberg and Settele (2016), question the monetary valuation of non-market goods and propose to focus economics on measuring only real things.

Bagstad et al. (2013) compared different decision support tools for valuing ecosystem services. Some of the most widespread public domain models, such as InVEST and ARIES, quantify services and their trade-offs at the landscape scale to support scenario analysis using biophysical units to which per-unit monetary values can be applied. They typically quantify ecosystem services using tables of coefficients for each land cover type derived from field experiments. Vigerstol & Aukema (2011) also compared the ecosystem services tools InVEST and ARIES with traditional hydrological tools that require more processing to assess ecosystem services.

Kang et al. (2022) stated that despite many ecosystem service valuation studies, calculated values presented wide variations and discrepancies. Their study showed that the highest value among the significant ecosystems is wetlands. That regulation of water flow services is of higher value than the rest of the ecosystem services. They divided valuation methods into eight categories, grouped into two types: the equivalent factor method group and the non-equivalent factor method group.

Within the non-equivalent factor group or primary data-based approaches, seven categories were included (TEEB, 2010; Zhang & Lu, 2010): market price method, shadow price method, avoided cost method, replacement cost method, travel cost method, contingent valuation and choice experiment methods, and others.

On the other hand, the equivalent factor method refers to ecosystem services calculated based on the relative weight of a particular ecosystem service compared to the standard (equal factor per unit area). In the study by Xie et al. (2017), the measure was the natural grain output from 1 ha of farmland. However, using a non-provisioning ecosystem service as the standard makes more reasonable this approach. Using a provisioning ecosystem service may not be feasible in certain areas, and using different services will result in various equivalent factors per unit, making the results non-comparable.

As in the case of most decision support tools for ecosystem service valuation, non-provisioning ecosystem services will be quantified in RECOVERY using tables of coefficients for each land cover type derived from field experiments (Bagstad et al., 2013).

With this starting point, we will explore ecosystem services valuation employing preference programming through approximate ratio comparisons, a development based on the analytic hierarchy process (Saaty, 1980) but with substantial practical potential due to the interactivity of its decision support that only requires linear programming to compute the results (Salo & Hämäläinen, 1995). This method allows ambiguous preference statements in hierarchical weighting, reducing the preference elicitation effort. Once a reference attribute is selected, the rest of the attributes will be compared relative to the reference attribute.

To achieve uniformity, the monetisation of all non-provisioning ecosystem services will be developed in RECOVERY based on the monetary valuation of carbon sequestration using the EU Emissions Trading System (2015). This trading system makes the regulatory ecosystem service of carbon sequestration the most direct and market-related valuation of non-provisioning services possible.

This methodology tries to avoid the lack of homogeneity caused by the current approaches. Using different valuation methods for non-provisioning ecosystem services generates a lack of uniformity in the valuation process that MCDA can hardly correct.

3 Valuation methodology

While provisioning ecosystem services will be valued according to market prices, non-provisioning ecosystem services will be quantified before their monetisation using tables of coefficients for each land cover type derived from field experiments, following Bagstad et al. (2013).

The valuation of the provisioning ecosystem services and the investments and costs incurred for any ecosystem services analysed will be done by calculating their net present value (NPV) over a sufficiently long period. A horizon of 70 years or more will be used to consider the residual value equal to zero. It is then necessary to define the discount rate used in the calculations. The selection of the appropriate discount rates and the subsequent analyses will be addressed in *D4.3 Adequate discount rates*.

On the other hand, the method selected to transform non-provisioning ecosystem services values into a common metric, an index between one and ten, will be local scaling. Local scaling sets upper and lower bounds using locally measured performance values instead of global scales that may cause irrelevance of differences between local measures.

Thus, all criteria performance values will have the same influence on the final scores of the alternatives if they are weighted equally (Martin & Mazzota, 2018), which is not going to be the case.

To monetise non-provisioning ecosystem services, implementation of techniques based on the propagation of imprecise preference statements in hierarchical weighting (Salo and Hämäläinen, 1995) will be used, employing the WINPRE (Workbench for Interactive Preference Programming) Program (Hämäläinen and Helenius, 1998).

Hierarchical weighting allows preference statements to be ambiguous, thus reducing the preference elicitation effort. Once a reference attribute has been selected, the remaining attributes are compared to the reference, considering the specific environment and the local scale used.

To achieve consistency, monetisation of all non-provisioning ecosystem services will build on the above comparison and the monetary valuation of the attribute with the most direct and market-related valuation possible: carbon sequestration, which was valuated using the EU Emissions Trading System (2015).

This trading system is the world's first primary carbon market. It remains the most important cornerstone of the EU's policy to combat climate change and its essential tool for reducing greenhouse gas emissions cost-effectively.

4 The Figaredo mining area

The Figaredo mining area will exemplify the valuation of ecosystem services provided by alternative ecological restoration scenarios. The aim is to estimate their contribution to human well-being, understand the incentives faced by decision-makers to manage ecosystems in different ways, and assess the values of alternative solutions.

The CLC classes that will be considered for the Figaredo mining area will be (Figure 4-1): Discontinuous urban fabric (112), Industry or commercial units (121), Mineral extraction sites (131), Dump sites (132), Pastures (231), Broad-leaved forest (311), Moors and heathland (322), Transitional woodland/shrub (324).

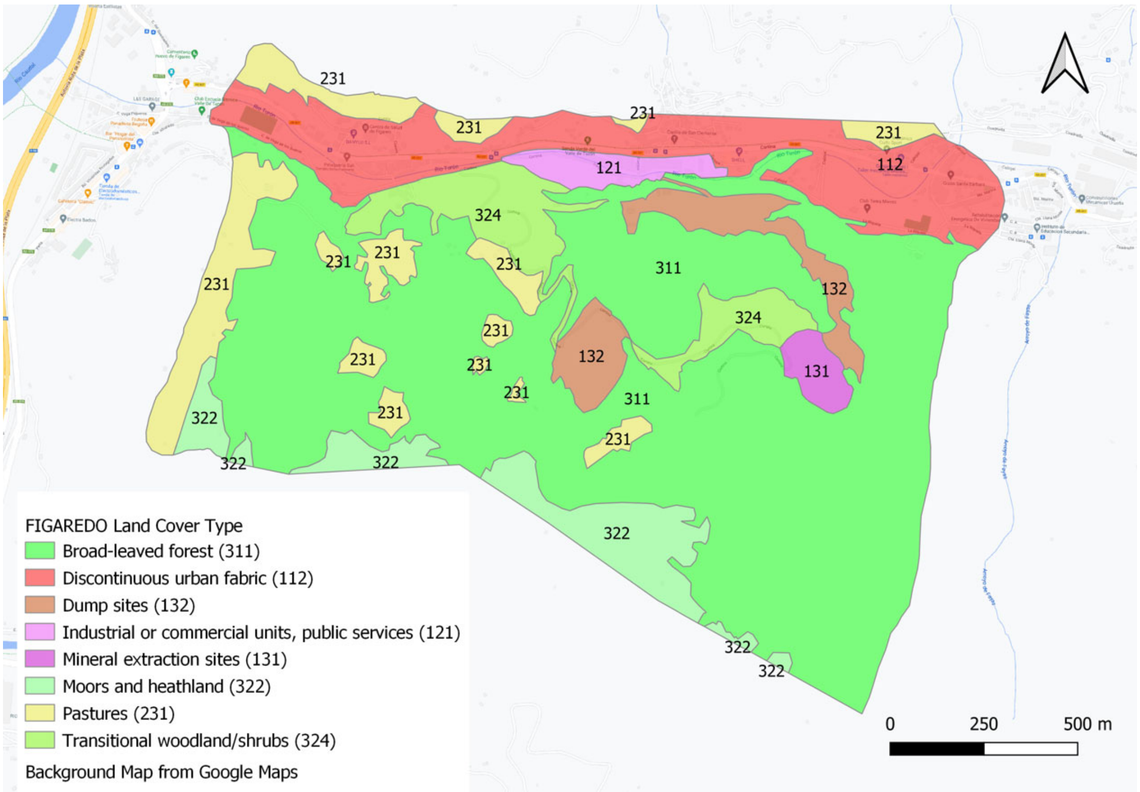

Figure 4-1. Presentation of the GIS of the CLC classes at the Figaredo mine.

For the selection of the alternative scenarios, the characteristics of the Figaredo area were taken into account, as well as the proposals obtained through a stakeholder consultation within the RECOVERY Project: (1) production of wood as raw material; (2) meat production; (3) broad-leaved forest, similar to those already present in the landscape of the region; (4) land use for renewable energy; (5) self-recolonisation; and (6) land use for physical recreation.

As wood production as raw material will be considered, a new land cover should be added to the previous list: Coniferous forest (312), which is not currently present in the study area.

Three of these scenarios were discarded for different reasons. The land use for renewable energy was not feasible due to the slopes of the waste heaps in the area and the northern orientation. Self-recolonisation was also not advisable because unrestored areas at the Figaredo mine did not achieve spontaneous revegetation after eleven years without appropriate land soil management (Figure 4-2).

Figure 4-2. The unrestored area near sector one was mined before 2009.

On the other hand, self-generated woody species cannot be compared in terms of productivity and profitability with, for example, pine plantations for timber production, which are widely used in Asturias. Finally, land use for physical recreation was ruled out, as there are numerous recreational facilities related to coal mining in the former coal mining area of Asturias.

Considering the specific features of the study area and the region in which it is located (Asturias, Spain), the proposed scenarios, as well as the results from D4.1 Suitable indicators, nine ecosystem services were selected for the Figaredo mining area following the CICES V5.1 classes: fibre production, food production, climate regulation (temperature), climate regulation (humidity), water flow regulation, erosion control, air purification, carbon sequestration, and biodiversity.

5 Ecosystem services quantification

As fibre production and food production are provisioning services, their specific valuation will be addressed in *D4.3 Adequate discount rates* and the costs incurred for any ecosystem service analysed.

5.1 Regulating services: climate regulation (temperature)

The air temperature was declared as the most apparent/suitable indicator when Schwarz et al. (2011) assessed the climate impact of different planning policies in the urban area of Leipzig in Germany, as trees and green regions moderate the climate. The corresponding CICES V5.1 code is 2.2.6.2, and the class 'Regulation of temperature and humidity, including ventilation and transpiration'. As air temperature is not easy to estimate spatially, thermal emissions from the earth's surface, which indicate the amount of energy emitted by bodies, could be used to measure temperature regulation.

In this case, the ecosystem service indicator could be land surface thermal emissions from the Landsat 7 ETM+ satellite (band 6) and the quantification method, the emission index, as used by Schwarz et al. (2011) but with the broad-leaved forest as the reference because its emission value is the lowest. Values were normalised in an index between 1 (highest emission) and 10 (lowest emission), according to equation (1), similar to that used by Larondelle & Haase (2012).

$$Index[i] = (max_{norm} + min_{norm}) - \left[(i - min) \times \frac{max_{norm} - min_{norm}}{max - min} + min_{norm}\right] \quad (1)$$

The thermal emissivity of the land cover and the respective normalised emission index adapted from Schwarz et al. (2011) are presented in Table 5-1.

Table 5-1. Thermal emissivity, evapotranspiration potential and runoff for CLC classes.

	Thermal emissivity		Evapotranspiration potential		Runoff	
CLC classes	Emission	Index	f	Index	% rainfall	Index
Discontinuous urban fabric (112)	139.4	3.5	0.9	2.8	65.0	1.0
Industry or commercial units (121)	141.5	1.0	0.8	1	65.0	1.0
Mineral extraction sites (131)	137.0	6.4	1.0	4.6	12.3	8.3
Dump sites (132)	139.0	4.0	1.0	4.6	12.3	8.3
Pastures (231)	135.4	8.3	1.1	6.4	0.6	9.9
Broad-leaved forest (311)	134.0	10.0	1.1	6.4	0.1	10.0
Coniferous forest (312)	137.4	5.9	1.3	10	6.2	9.2
Moors and heathland (322)	137.0	6.4	1.1	6.4	12.3	8.3
Transitional woodland/shrub (324)	136.0	7.6	1.1	6.4	0.2	10.0

Sources of uncertainty in this assessment are the differences in values under different climatic conditions, as these values were obtained for the urban region of Leipzig.

5.2 Regulating services: climate regulation (humidity)

Humidity (evapotranspiration) was selected by Schwarz et al. (2011) as a second indicator for estimating local climate regulation, as forests and green areas influence precipitation and water availability both locally and regionally. Evapotranspiration is the sum of the evaporation of water from the land surface and transpiration from vegetation.

While CICES V5.0 shares in code 2.2.6.2 both temperature and humidity regulation, the old version V4.3 had different codes for them: 2.3.5.2 'Micro and regional climate regulation', and 2.2.3.2 'Ventilation and transpiration'. The reason is that the classification structure of provisioning services in V4.3 was changed in V5.1 to allow aggregation when the end-use is unknown. The classification can be more easily used for accounting purposes.

Although there is a linear relationship between evapotranspiration and latent heat of vaporisation (the higher the evapotranspiration, the lower the energy available as sensible heat), this correlation disappears when analysing the total thermal emissivity. Thus, splitting the two services would facilitate the analysis. In this case, the ecosystem service indicator could be the evapotranspiration potential, as Schwarz et al. (2011) used.

The quantification method will approximate the evapotranspiration potential of the different land cover classes. Schwarz et al. (2011) used equations based on empirical estimates and considered soil types and climatic conditions. The evapotranspiration potential f[i] was calculated according to equation (2).

$$f[i] = (max\ evapotranspiration\ [i] \div ET_0) \tag{2}$$

where ET_0 is the reference evapotranspiration potential of the 12 cm tall grass.

Values were again normalised between 1 (lowest evapotranspiration potential) and 10 (highest evapotranspiration potential). Equation (3) was used, as it was unnecessary to reverse the ranking to reflect the lowest evapotranspiration as the highest index.

$$Index[i] = \left[(i - min) \times \frac{max_{norm} - min_{norm}}{max - min} + min_{norm}\right] \tag{3}$$

The evapotranspiration potential, adapted from Schwarz et al. (2011), and the respective normalised emission index are presented in Table 5-1. Sources of uncertainty in this assessment are again differences in soil types and values under different climatic conditions, as these values were obtained for the urban region of Leipzig.

5.3 Regulating services: water flow regulation

Water flow regulation is another regulating service, as Asturias is a region with high rainfall. The corresponding CICES V5.1 code is 2.2.1.3, and the class 'Hydrological cycle and water flow regulation'.

The ecosystem services indicator could be the volume of water retained by vegetation per ha, and the quantification method is the statistical runoff estimated by Nunes et al. (2011).

Some approximations had to be considered, as not all CLC classes of Figaredo mines were presented in Nunes et al. (2011). The values of the rainiest year between the two years analysed (2006) were selected, and the mineral extraction sites and dump sites were assimilated to afforested land. The value chosen for coniferous forests was the mean between broad-leaved forest, moors, and heathland.

According to Tanouchi et al. (2019), the range of the impervious surface ratio of the discontinuous urban fabric is between 50% and 80%, so a mean runoff value of 65% of the total rainfall was assigned to both the discontinuous urban fabric and industry or commercial units. The quantification results are presented in Table 5-1, and a water flow regulation index is calculated according to equation (1).

The assessment's sources of uncertainty will be the different values in different climatic environments/conditions and assumptions based only on one year's rainfall.

5.4 Regulating services: erosion control

Erosion control is also a regulating service to be considered, although its importance in the Asturias region is not very significant. The corresponding CICES V5.1 code is 2.2.1.1 and the class 'Control of erosion rates'. The ecosystem services indicator could be the soil erosion in g/m_2, and the quantification method the statistical runoff as estimated by Nunes et al. (2011).

Using the same assumptions as with water flow regulation and values from the same year (2016), Table 5-2 presents soil erosion in g/m_2 and an erosion control index calculated according to equation (1). In the case of the discontinuous urban fabric and industry or commercial units, as the non-impervious surface, according to Tanouchi et al. (2019), was 35%, this percentage was used to calculate their soil erosion based on that of mineral extraction sites and dump sites.

Table 5-2. Soil erosion, dry deposition of pollutants and above-ground carbon storage for the different CLC classes.

	Soil erosion		Dry deposition of pollutants		Above-ground carbon storage	
CLC classes	**g/m^2**	**Index**	**k /year**	**Index**	**t C/ha**	**Index**
Discontinuous urban fabric (112)	193.0	6.9	2.02	1.0	20.0	3.6
Industry or commercial units (121)	193.0	6.9	2.02	1.0	8.52	2.1
Mineral extraction sites (131)	551.3	1.0	2.02	1.0	≈ 0	1.0
Dump sites (132)	551.3	1.0	2.02	1.0	≈ 0	1.0
Pastures (231)	2.4	10.0	149.4	6.2	≈ 0	1.0
Broad-leaved forest (311)	1.4	10.0	258.9	10.0	68.31	10.0
Coniferous forest (312)	15.6	9.6	258.9	10.0	≈ 0	1.0
Moors and heathland (322)	29.8	9.1	120.2	5.1	4.0	1.5
Transitional woodland/shrub (324)	1.2	10.0	189.6	7.6	10.12	2.3

The assessment's sources of uncertainty will be the different values in different climatic environments/conditions and assumptions based only on one year's rainfall.

5.5 Regulating services: air purification

Plants provide air purification or removal of air pollution. They have large surface areas for particle deposition and adsorption of gases by the leaf or chemical reactions on the leaf surface. These processes are often referred to as 'dry deposition'. The amount of pollution removed by plants depends on their leaves' size and surface area but can vary depending on climate, time of year, and other atmospheric pollutants.

The CICES V5.1 code is 2.2.6.1. The class is "Regulation of chemical composition of atmosphere and oceans". The ecosystem service indicator could be pollutant capture. The quantification method could be dry deposition of the following pollutants, as used by Jones et al. (2017): Sulphur dioxide (SO2), coarse particulate matter (PM10), fine particulate matter (PM2.5), ammonia (NH3), nitrogen dioxide (NO2) and ozone (O3). Other interesting studies consider CO (Nowak et al., 2006), but the variations should not be significant as the pollutants will be considered together.

Table 5-2 presents the dry deposition of pollutants by land cover classes adapted from Jones et al. (2017) and a pollutant dry deposition index calculated according to equation (3).

Again, sources of uncertainty in the assessment will be the different values in different climatic and geographical environments/conditions.

5.6 Regulating services: carbon sequestration

Carbon sequestration was the last regulating service considered. In the case of pastures and coniferous forests, since they are considered provisioning services, this is incompatible with accounting for carbon sequestration as a regulating service. The CICES V5.1 code will be again 2.2.6.1, and the class “Regulation of chemical composition of atmosphere and oceans”.

The ecosystem services indicator shall be above-ground carbon storage/ha. The above-ground carbon storage quantification method will be linked to land use in t C/ha, as Strohbach and Haase (2012) estimated in a study on above-ground carbon storage in Leipzig (Germany).

Table 5-2 presents the above-ground carbon storage per land cover to be considered, adapted from Strohbach & Haase (2012), and a carbon storage index calculated according to equation (3).

In this case, an indirect monetary valuation of the ecosystem service is possible using the EU Emissions Trading System (2015). Sources of uncertainty in the assessment are the values at different locations, as these values were obtained for Leipzig.

5.7 Cultural services: qualities of species or ecosystems (biodiversity)

The qualities of species or ecosystems (biodiversity) or biophysical features (landscapes) representing typical Asturian forests (Broad-leaved forests) in the Figaredo mine area was the last ecosystem service to be analysed.

The CICES V5.1 code is 3.2.2.1, and the class ‘Characteristics or features of living systems that have an existence value’. An example of service should be ‘areas designated as wilderness, and the ecosystem services indicator could be the type of living systems or environmental settings.

The quantification method could be the number of endemic or quasi-endemic species observations. This particular ecosystem service represents an excellent proxy for quantifying biodiversity. Code 3.2.2.2 has the same ecosystem service class and the same indicator. The only difference is that while the simple descriptor of this code is ‘things in nature that we want future generations to enjoy or use’, the first code was ‘the things in nature that we think should be conserved’. In our view, the two are complementary and indissoluble, at least in this case.

Although there are different metrics to assess biodiversity, considering aspects such as species richness, evenness and identity, for the specific biotope of the Figaredo mine, a study on the nexus between urban shrinkage and ecosystem services by Haase et al. (2014) could be used as a reference to simplify the process.

Table 5-3 presents the impact on the biodiversity of the different land cover cases in the Figaredo mine area, adapted from Haase et al. (2014) and the biodiversity index calculated with equation (3).

Table 5-3. Biodiversity impact and respective normalised impact index (adapted from Haase et al., 2014).

CLC classes	Impact	Index
Discontinuous urban fabric (112)	0	1
Industry or commercial units (121)	0	1
Mineral extraction sites (131)	1	4
Dump sites (132)	1	4
Pastures (231)	2	7
Broad-leaved forest (311)	3	10
Coniferous forest (312)	2	7
Moors and heathland (322)	2.5	8.5
Transitional woodland/shrub (324)	2.5	8.5

According to Cavard et al. (2011), different tree species, as in a typical broad-leaved forest in the Figaredo mine (mixed forest), are associated with a more prominent diversity provision than in a case of a single stand forest of conifer plantations. In addition, as conifer plantations will be used for fibre production, their impact on biodiversity was considered at the same level as pastures. On the other hand, moors and heathland and transitional woodland/shrub have been impacted midway between pastures and broad-leaved forests.

Finally, Table 5-4 summarises the ecosystem service indicators considered important/relevant in the Figaredo mine area, their quantification methods and the primary references used.

Table 5-4. Summary of non-provisioning ecosystem service indicators, quantification methods and primary references used in the Figaredo mine case study.

Ecosystem service	Indicator	Quantification method	References
Climate regulation (Temperature)	Land surface thermal emissions	Thermal emissivity	Schwarz et al. (2011)
Climate regulation (Humidity)	Evapotranspiration	Evapotranspiration potential	Schwarz et al. (2011)
Water flow regulation	Runoff	Runoff in % of total rainfall	Nunes et al. (2011)
Erosion control	Soil loss	Soil erosion in g/m^2 during a monitored period	Nunes et al. (2011)
Air purification	Pollutant capture	Dry deposition of pollutants in t/year	Jones et al. (2017)
Carbon sequestration	Carbon storage	Above-ground carbon storage in t/ha	Strohbach & Haase (2012)
Qualities of species or ecosystems (Biodiversity)	Impact of shrinkage-related cover patterns	Degree of suitability	Haase et al. (2014)

6 Estimating the ecosystem services provision

To estimate the ecosystem services provision of each proposed scenario, the application of techniques based on the propagation of imprecise preference statements in hierarchical weighting (Salo and Hämäläinen, 1995) using the WINPRE program (Hämäläinen and Helenius, 1998) was used.

It was then first necessary to select a reference attribute. Biodiversity was chosen as the reference attribute because, of all the attributes, it was the one that allowed comparisons to be made with the others in the most obvious way, which facilitated the development of the process. The rest of the attributes were then compared with the reference attribute.

Rank orderings should not change with a different anchor, as they are bi-univocal among the various ecosystem services. What may change is only the difficulty of establishing these rank orderings. That is why biodiversity was the anchor selected, as it is the most intuitive among them. Figure 6-1 presents the results of the comparison carried out using the Delphi method and the WINPRE program, developed by experts from Hulleras del Norte, S.A. (Spain), the School of Mining, Energy and Materials Engineering of Oviedo (Spain), and the Central Institute of Mining in Katowice (Poland).

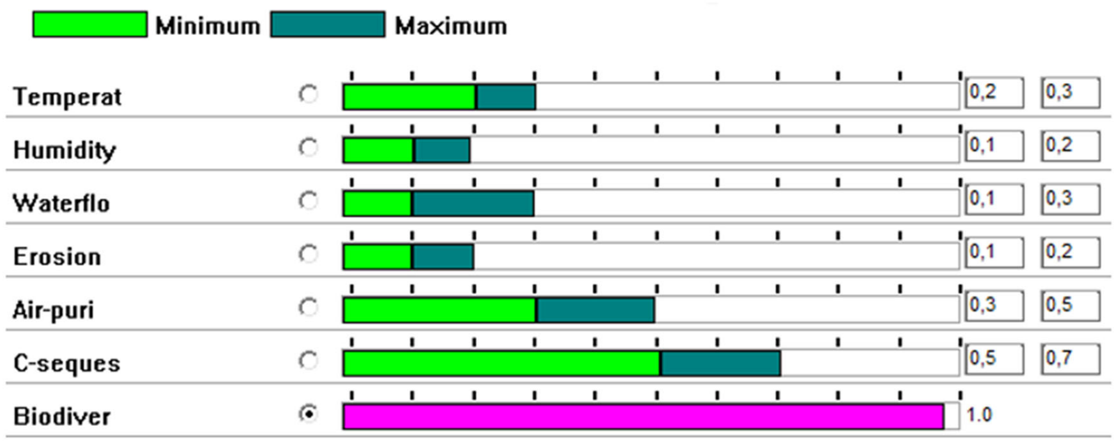

Figure 6-1. Comparison of attributes (ecosystem services) to the reference attribute (biodiversity).

While biodiversity was rated as the benchmark with 100% importance, in the case of humidity and erosion, their importance was ranked between 10 and 20% of biodiversity. The Asturias region is humid, and erosion is not a problem except for steep slopes. Carbon sequestration was considered between 50 and 70% of biodiversity importance, and so on. No attribute was given more than 100% importance, although this may be the case in other comparisons.

The second step consisted of giving values to the different scenarios for each attribute. Value ranges were assigned because the calculated indices cannot be considered entirely accurate. Figure 6-2 presents the value ranges for the temperature attribute derived from Table 2. When an index is scored with decimals, the selected value range was between the figure's lower and upper integer values, e.g., the Fibre index was 5.9 (coniferous forest), and the Food index was 8.3 (pastures). When an index has an integer value, the values selected range between that value and one point less, e.g., the Landscape index is 10 (broad-leaved forest).

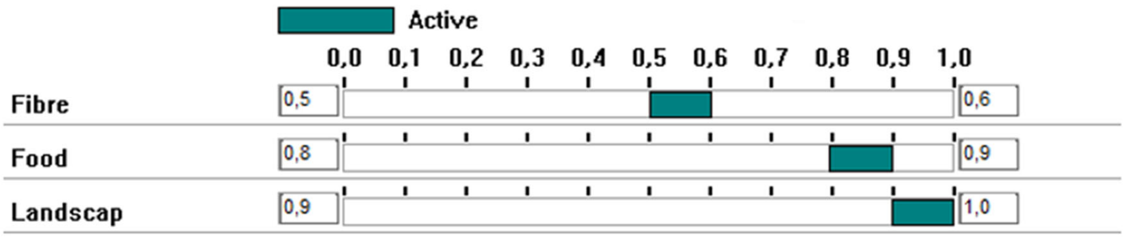

Figure 6-2. The ranges of values for attribute temperature.

Following the calculations developed with WINPRE, Table 6-1 presents the pairwise comparisons with upper and lower bounds for the alternative scenarios.

Table 6-1. The ranges of values for the different scenarios.

Scenarios	Lower bound	Mean	Upper bound
Landscape	0.87	0.93	0.99
Fibre	0.49	0.60	0.71
Food	0.47	0.57	0.67

The ecosystem service for which valuation is most feasible must first be selected to monetise the set of ecosystem services. In this case, the indirect monetary valuation of carbon sequestration through the EU Emissions Trading System (2015) is the most feasible.

According to the EU Emissions Trading System (2015), between 2019 and 2020, the average value of EU Allowances, which allows for the emission of 1 tonne of carbon dioxide equivalent, was about 25 €/t (Figure 6-3).

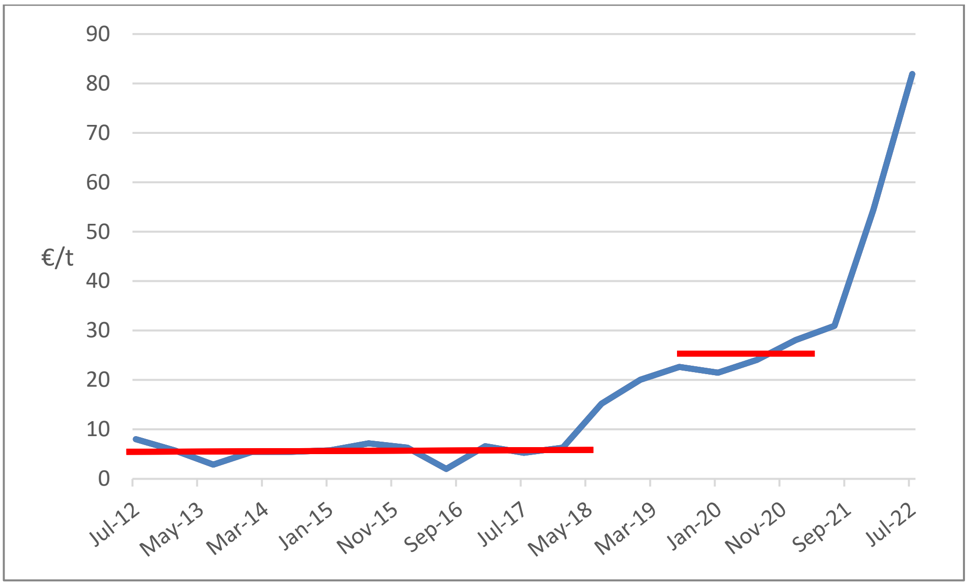

Figure 6-3. EU Carbon Permits (€/t). Adapted from EMBER (2021) and www.tradingeconomics.com.

As 3.67 t CO_2 contain 1 t C, the average value of sequestration of 1 t C can be estimated at 91.75 €/t. Therefore, an above-ground carbon storage rate of 10.0, equivalent to 68.31 t C/ha (Table 3), should be valued at 6,267 €/ha. This value will be the reference value for 100% weighted ecosystem services. We assume that all non-provisioning ecosystem services weighted at 100% are worth the same, given that the specific values for each ecosystem service will come from the relative comparison between them.

Table 6-2 shows the current valuation of ecosystem services, with biodiversity being the only attribute valued at 100% and used as the reference attribute. Thus, it was given a value of 6,267 €. Finally, the total value of the highest possible contribution of ecosystem services in the Figaredo mine area is 16,294 €/ha.

Table 6-2. The current valuation of ecosystem services per ha.

Attribute	Comparative average weight*	Ecosystem service value per ha
Temperature	25%	1,567 €
Waterflow	20%	1,235 €
Erosion	15%	940 €
Air purification	40%	2.507 €
Carbon sequestration	60%	3,760 €
Biodiversity	100%	6,267 €
Total		16,294 €

* Comparison of other attributes (ecosystem services) concerning the reference attribute (biodiversity), as presented in Figure 6-1.

No discount should be applied to the ecosystem service values in Table 6-2, as they do not represent real cash flows but timeless values.

Finally, Table 6-3 presents the total value of the different scenarios per ha using the average percentage of the ranges of values shown in Table 6.

Table 6-3. The ecosystem services value of the different scenarios per ha.

Scenarios	Mean	E.S. value
Landscape	0.93	15,154 €
Fibre	0.60	9,777 €
Food	0.57	9,288 €

7 Conclusions and lessons learned

In this deliverable, a new valuation methodology for non-provisioning ecosystem services was developed, as evidence was found regarding the lack of homogeneity caused by the existing approaches.

The valuation of provisioning ecosystem services as well as the valuation of the investments and costs related with non-provisioning ecosystem services will be addressed in *D4.3 Adequate discount rates*, as it will be necessary to discount these ecosystem services over an adequate period.

The developed methodology was based on the implementation of techniques based on the propagation of imprecise preference statements in hierarchical weighting, as well as on the monetary valuation of the attribute with the most direct and market-related valuation possible.

The lessons relevant to RECOVERY from the valuation of non-provisioning ecosystem services can be summarised as follows:

1. Multi-criteria decision analysis was developed to determine the best choice based on the scores of different criteria and the relative weights given to those criteria. However, it is complicated to assign relative weights to criteria that have been evaluated with varying assessment methods.

2. Non-provisioning ecosystem services have to be quantified in the first place using tables of coefficients for each land cover type derived from field experiments. Then, they will be transformed into a standard metric employing local scaling instead of global scales that may cause irrelevance differences between local measures.

3. The source of uncertainty in these valuations will be the different values in different climatic environments/conditions and assumptions based on specific areas or circumstances.

4. Then, the implementation of techniques based on the propagation of imprecise preference statements in hierarchical weighting may be used, with a reference attribute having to be selected to compare the remaining attributes. Biodiversity was selected as, of all the attributes, it was the one that allowed comparisons to be made with the others in the most obvious way.

5. Finally, monetisation of all non-provisioning ecosystem services should be built on the above comparison and the monetary valuation of the attribute with the most direct and market-related valuation possible: carbon sequestration using the EU Emissions Trading System.

6. As the average value of EU Allowances during 2019 and 2020 was used, variation in the value may change the results. This question will be discussed in *D4.3 Adequate discount rates.*

8 Glossary

CAPM: Capital asset pricing model

CBA: Cost-benefit analysis

CICES: Common International Classification of Ecosystem Services

CLC: Corine Land Cover

EU: European Union

EURIBOR: Euro Interbank Offered Rate

MCDA: Multi-criteria decision analysis

NPV: Net present value

TEEB: The Economics of Ecosystems and Biodiversity

TEV: Total Economic Value

WINPRE: Workbench for Interactive Preference Programming

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