

Pilot testing of the SEEA-EEA Framework in Mexico

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NCAVES project report

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1. Introduction

The Natural Capital Accounting and Valuation of Ecosystem Services Project is a three-year (2017-2019), global project launched jointly by United Nations Statistics Division (UNSD), United Nations Environment Programme, and the United Nations Convention on Biological Diversity (CBD), with financial support from the European Union. Brazil, China, India, Mexico, and South Africa were chosen as the initial countries to pilot test the project.

The project aims to advance the knowledge agenda on environmental-economic accounting, particularly ecosystem accounting, by initiating pilot testing of the *UN System of Environmental Economic Accounts-Experimental Ecosystem Accounting* (SEEA-EEA; see UN, 2014b) framework in select countries where biodiversity is at stake, with a view to:

- Improving the measurement of ecosystems and their services (both in physical and monetary terms) at the national/subnational level;
- Mainstreaming biodiversity and ecosystems in national/subnational level policy-planning and implementation;
- Contribute to the development of internationally agreed methodology and its use in partner countries.

The Natural Capital Accounting and Valuation of Ecosystem Services in Mexico (NCAVES-Mexico) project was officially launched in June 2017 and implementation began in September 2017 when the project consultant was contracted. The Programme of Work 2018 for the NCAVES-Mexico project was submitted to and approved by the Interinstitutional Working Group on Experimental Ecosystem Accounting in February 2018; it included two major tasks:

- conducting a <u>country assessment</u> of natural capital accounting and valuation of ecosystem services in Mexico, based on desk study and visits to relevant organisations. The assessment report would serve as an input to develop a national plan for advancing environmental-economic and ecosystem accounting in Mexico, and
- compiling <u>pilot accounts</u> in physical and, whenever possible monetary terms— of select ecosystem services based on existing data and using the UN-SEEA-EEA framework.

The *Country Assessment on Natural Capital Accounting and Valuation of Ecosystem Services in Mexico* was completed in December 2018. The report describes the current situation in Mexico with regard to: a) environmental-economic accounting; b) valuation of ecosystem services; c) mechanisms for payment for environmental services; d) inclusion of ecosystem services concepts in public policy, planning and regulatory instruments; and e) the country's participation in international initiatives for valuing ecosystem services. The report also identifies opportunity areas where adoption of the SEEA-EEA framework in Mexico might be useful. The report is currently being edited for separate publication.

In this document I describe the data sources, data compilation processes, and analytical methods used in, as well as the main findings from, the pilot studies carried out under the NCAVES-Mexico project. Some recommendations on how to improve the quality and scope of ecosystem service accounts in the medium term are also provided. Finally, the potential relevance and influence of the results from the pilot studies and, more generally, of adopting and implementing the SEEA-EEA framework for valuing Mexico's natural capital and ecosystem services is discussed.

Technical work for the pilot studies was carried out in close collaboration and consultation with personnel from INEGI's Dirección de Recursos Naturales. In addition, data and technical assistance for specific themes was kindly provided by experts from Dirección General de Estadística e Información



Ambiental-SEMARNAT (surface water supply, Human Footprint Index) and Gerencia de Monitoreo Forestal - CONAFOR (forest carbon storage and sequestration). Technical exchange meetings were carried out in July-September 2019 to present the results, source data, and methods developed/used in the pilot studies to the consideration of technical experts from CONAFOR, Servicio de Información Agroalimentaria y Pesquera–SADER and Comisión Nacional del Agua. They all concurred with the data and methodological approaches used in the pilot studies and found their results reasonable; they also provided additional recommendations for further work on ecosystem services accounting in Mexico.

Finally, given the still experimental nature of the SEEA-EEA framework, its strict methodological requirements, as well as its very recent —and, therefore, limited — application in real-world cases, it is not surprising that challenges are found in various stages of the implementation process. For these reasons, a revision process of the SEEA-EEA conceptual and methodological framework was launched in 2018 (UN, 2018). Early results from the revision process were recently (May-June, 2019) released, providing further guidance and clarification on these topics. Unfortunately, work referred to in this report was carried out during 2018 and the first half of 2019 so that timing precluded fully incorporating the new developments from the revision process into the Mexico pilot studies. Nevertheless, in this document I try to frame —as much as feasible— the issues found and the decisions made in relation to the results from the revision process.

2. The SEEA-Experimental Ecosystem Accounting framework

Economic and other human activities have been causing an overall degradation of ecosystems, thus reducing the ecosystems' capacity to maintain their life-supporting structure and processes. Ecosystem accounts can provide valuable information for tracking changes in ecosystems and for linking those changes to human activities.

The *SEEA-Experimental Ecosystem Accounting* is a relatively recent development within environmental-economic accounting; it complements and expands on the *UN SEEA-Central Framework* (UN, 2014a). While the latter starts from the perspective of the economy and incorporates relevant environmental information on natural inputs, residual flows and environmental assets, the SEEA-EEA starts from the perspective of ecosystems and links ecosystems to economic and other human activities. Together these two approaches provide the potential to fully describe the relationship between the environment, the economy and other human activities.

In the SEEA-EEA framework, the environment and ecosystems are not simply seen as sources of materials and a sink for waste streaming from economic and other human activities, but as tangible assets that provide materials such as food, wood, fibres, etc. and maintain essential processes such as water capture and filtration, capture and sequestration of CO₂, etc. The SEEA-EEA framework utilizes a stocks-and-flows model in which ecosystems are conceptualized as individual assets existing within spatially defined areas at a given point in time (UN, 2014b). Assets can be identified in a spatially explicit manner and evaluated in terms of their extent (*i.e.*, size of the stock) and condition (and other relevant characteristics such as ownership, management, etc.). Ecosystem assets deliver flows that are then combined with human inputs (*e.g.*, capital and labour) to produce goods and services that can benefit people —that is, ecosystem services.

The SEEA-EEA framework encompasses all areas —even highly anthropized areas such as crop fields, pasturelands and human settlements, as these also deliver ecosystem services. Unlike other more ecologically oriented approaches (*e.g.*, the Millennium Ecosystem Assessment's, in which ecosystem services are directly defined as the benefits that people obtain from ecosystems), in the SEEA-EEA framework a narrower definition of ecosystem services is adopted (*viz.*, "...the contributions of



ecosystems to benefits used in economic and other human activity..."), in order to recognize —and separately account for— the human inputs required for turning raw materials/services into actual goods and products that people use. In addition, in this framework ecosystem services are classified into just three broad categories: Provisioning, regulating, and cultural services. Supporting services (*e.g.*, soil formation, oxygen production through photosynthesis, pollination, etc.), which are explicitly recognized in more ecologically-oriented frameworks (*e.g.*, Millennium Ecosystem Assessment, IPBES), are excluded as these are seen as intermediate services that only indirectly (or over a very long time) impact people (and also in order to avoid double-counting the contribution of ecosystems to the generation of benefits; UN, 2014b).

Ecosystem assets and services can be measured in physical terms and in monetary terms. By measuring changes in the extent and condition of ecosystems and in their ability to provide ecosystem services one can, for example, understand whether economic activities are sustainable or are diminishing the capacity of ecosystems to provide those services in the future (UN, 2014b). The purpose of the SEEA-EEA framework is to measure and monitor such changes over time, in a spatially explicit manner, and in a way that is consistent with economic national accounting. Conceptually, the ecosystem accounts constitute an integrated representation of the environmental and economic characteristics of the study area, describing (in terms of extent and condition) the ecosystem assets existing therein, together with the use of such assets by people (in the form of ecosystem services, goods and products).

On the methodological side, SEEA-EEA aims to provide a measurement framework for integrating biophysical data, assessing ecosystem services, tracking changes in ecosystems and their services, and linking those changes to economic and other human activity, using a spatially explicit approach. The *SEEA-Experimental Ecosystem Accounting* document (UN, 2014b) and the *Technical recommendations in support of the SEEA-EEA* (UN, 2017) provide methodological guidelines and recommendations to test and experiment on ecosystem accounting issues. The overall workflow for valuing ecosystem services within the SEEA-EEA framework is schematically described in Figure 1 below. In this framework, the capacity of a given ecosystem to supply a given service is a function of both, the area it covers (*i.e.*, its extent) and its condition (*i.e.*, its quality):

Supply of ecosystem service = f(Ecosystem extent, Ecosystem condition)

Accordingly, in the workflow (Fig.1) ecosystems are first characterized in terms of both, quantitative (ecosystem extent) and qualitative (ecosystem condition) descriptors. Then, the supply and use of the services the ecosystem provides are measured in physical terms (green boxes in Fig.1) and, finally, valued in monetary terms (grey boxes) (see UN, 2017). First accounting for ecosystems in physical terms is a key feature of the SEEA-EEA framework, as the monetary valuation of the supply and use of ecosystem services is necessarily dependent on the availability of information in physical terms, as observable market values for ecosystems and their services are non-existent or very scarce (UN, 2014b). As described in the following sections, this workflow was followed for mapping and valuing —in physical terms only—select ecosystem services in Mexico.





Fig.1 Diagram showing the general workflow for valuing ecosystem services within the SEEA-EEA framework; steps shown in green colour are carried out in physical terms, whereas the ones shown in grey are formulated in monetary terms (modified from UN, 2017).

When applying the SEEA-EEA framework in practice, it is important to bear in mind that this does not yet constitute an international statistical standard. At this stage, the SEEA-EEA framework only provides an accounting framework for research and testing on ecosystems and their relationship to economic and other human activity; work in this field is still deemed as experimental (UN, 2014b, 2018). Admittedly, the *SEEA-Experimental Ecosystem Accounting* document (UN, 2014b) and the *Technical recommendations in support of the SEEA-EEA* (UN, 2017) do not yet provide definitive advice on how to address the several conceptual and methodological challenges faced when compiling ecosystem accounts in practice.

In fact, early experiences with the SEEA-EEA framework have shown (UN, 2017; Bogaart *et al.*, 2019) that, in a number of areas, further conceptual advancement is still necessary and that, in all areas, better measurement methods still need to be developed and/or tested. Because of this, in 2018 a revision process of the SEEA-EEA framework was launched (UN, 2018), topics demanding specific attention (*viz.*, spatial units, ecosystem condition, ecosystem services and valuation) were identified, and working groups established accordingly. As an early result from the revision process, a series of discussion papers were developed and recently released (*e.g.*, Bogaart *et al.*, 2019; Keith *et al.*, 2019; Maes *et al.*, 2019; Czúcz *et al.*, 2019). The on-the-ground, pilot tests of the SEEA-EEA framework being currently carried out in Mexico and other countries are expected to provide valuable lessons to make further progress in these topics.

3. Pilot testing the SEEA-EEA framework in Mexico

This report describes the data and analytical methods used in, as well as the main findings from, the pilot accounts of select ecosystem services in Mexico using the UN-SEEA-EEA framework. Several objectives were pursued through these studies:

- Firstly, we aimed to test whether —and to what extent— the SEEA-EEA approach for measuring and monitoring ecosystem services flows can be applied at both the country-wide and the State-level scale in Mexico. This entailed examining whether —and how— multiple ecosystem services (ES) can be spatially modelled and analysed using existing data and in a way suitable for ecosystem accounting purposes as per the SEEA-EEA framework.
- Secondly, we aimed to examine whether the results from the pilot accounts would yield information relevant to decision-making about natural resource management in Mexico, and how the results from



the pilot accounts may be used by government institutions involved in natural resource management; and

• Thirdly, identifying how the experience of modelling ecosystem services and compiling the accounts for Mexico's conditions can contribute to the on-going development of the SEEA Experimental Ecosystem Accounting framework.

The ecosystem services tested were chosen based on several considerations. First and foremost, we considered data availability (see section **4. General considerations on data and methods** below). Secondly, following the recommendations in the SEEA-EEA framework (UN, 2014b) and based on personal expert judgement/experience as well as feedback from key personnel at INEGI and SEMARNAT, we considered the country's policy priorities, the services' sensitivity to environmental changes and their socio-economic importance. Finally, we also considered the experience gained in a previous pilot test carried out under the ANCA project. This way, the following four ecosystem services were chosen for pilot testing the SEEA-EEA framework in Mexico both, at the State- and the countrywide level:

- Carbon sequestration
- Surface water supply
- Food crop production by agroecosystems
- Coastal protection by mangrove ecosystems

The SEEA-EEA framework recommends following the Common International Classification of Ecosystem Services (CICES 2018) to avoid ambiguity when identifying, classifying and naming ecosystem services. The four ecosystem services selected for the NCAVES-Mexico pilot studies correspond to provisioning (food crop production, water abstraction) and regulation (carbon sequestration and coastal protection) services in the CICES v.5.1 classification scheme (see Table 1 below). Notice that our study on surface water supply does not distinguish between water used for drinking or for non-drinking purposes.

Section	Division	Group	Class	Code	Class type
Provisioning (Biotic)	Biomass	Cultivated terrestrial plants for nutrition, materials or energy	Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes	1.1.1.1	Crops by amount, type (e.g. cereals, root crops, soft fruit, etc.)
Provisioning (Abiotic)	Water	Surface water used for nutrition, materials or energy	Surface water for drinking Surface water used as a material (non-drinking purposes)	4.2.1.1 4.2.1.2	By amount, type, source
Regulation & Maintenance (Biotic)	Regulation of physical, chemical, biological conditions	Atmospheric composition and conditions	Regulation of chemical composition of atmosphere and oceans	2.2.6.1	By contribution of type of living system to amount, concentration or climatic parameter
Regulation & Maintenance (Biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Hydrological cycle and water flow regulation (Including flood control, and coastal protection)	2.2.1.3	By depth/volumes

 Table 1. Ecosystem services evaluated in the NCAVES-Mexico project, classified according to the Common International Classification of Ecosystem Services v.5.1 (CICES 2018).



4. General considerations on data and methods

Applying the SEEA-EEA framework requires of various types of statistical and geographic data from different sources, as well as the use of a range of analytical and spatial modelling methods. Given its experimental nature, as well as its still limited application in real-world cases, no officially approved nor generally adopted data sources and methods are yet available for most of the stages of the process (Maes *et al.*, 2019). At the same time, this leaves the way open for potentially using any of a wide range of data sources and/or methodological approaches that could, in principle, seem useful or suitable for each stage of the workflow.

One of the main findings of the *Country Assessment on Natural Capital Accounting and Valuation of Ecosystem Services in Mexico* was that, in general, there is insufficient data/information on ecosystem services that is solid, readily available, easily accessible, and of sufficient temporal and spatial resolution. On the one hand, the only information currently available specifically dealing with the volume, flow and value of ecosystem services in Mexico is that compiled or produced by the many local-level study cases of ecosystem services valuation that have been conducted in the country over the last 25 yr. Such studies, however, are affected by several issues that prevent their results from being more generally useful and easily accessible. On the other hand, the assessment also showed that, as part of their operation, various government agencies regularly collect statistical and geographical data and information which are fully relevant and potentially useable for ecosystem services accounting, they need to be first reformatted, adapted, reformulated or used as inputs for biophysical models/analyses prior to use them for analysing ecosystem services stocks and flows, etc.

For these reasons, the selection of inputs (statistical and geographic data) and analytical and modelling methods to be used in the pilot studies of the NCAVES-Mexico project was based on the following principles:

- Use local, official data/information or, at least, data/information that is produced or used by official entities in the country;
- Use data with country-wide coverage;
- Use spatially explicit data of adequate resolution; given the explicit purpose of the NCAVES-Mexico project to conduct pilot tests both at the State- and countrywide level, suitable data should have a moderate to high spatial resolution (*i.e.*, 1:250,000 scale or higher; pixel size 250m or smaller);
- Use data sets encompassing a consistent time-series or that, at least, provide comparable data for various points in time (a few years);
- Use data that are freely, openly available;
- Use explicit, transparent and repeatable methods for analysis and modelling;
- Avoid using "black-box" methods or models.

With regard to data, in particular, the overall approach was to use publicly available data sources (*e.g.*, official websites, published studies and assessments, annual reports, etc.) and to adapt these as best as possible to fit the SEEA-EEA methodological requirements. In many cases, clarification or additional information was sought from primary data sources (for example, the National Forests Commission, the Ministry of Agriculture, INEGI, etc.). For easier reading, the main text of this document focuses on presenting the results and findings from the pilot studies. The data, data sources, data compilation processes, and analytical methods used in such studies for mapping and valuing (in physical terms) ecosystem services are described, in detail, in Annex B of this document. The specific issues related to mapping and assessing the extent and condition of Mexico's ecosystems, as well as the rationale



underlying the decisions made in the NCAVES-Mexico project, are discussed at length in Annex A.

5. Ecosystem classification

The first step in the SEEA-EEA framework (see Fig.1) involves delineating the ecosystems (or *ecosystem assets*) present in the study area (or *ecosystem accounting area*). In principle, a range of characteristics can be used for this purpose, including ecological (*e.g.*, vegetation type, soil type, hydrology, land management, etc.) and non-ecological (*e.g.*, land use, ownership, site features, etc.) features. Accordingly, many different approaches and systems for classifying ecosystems have been proposed and are in use worldwide. However, the UN-SEEA ecosystem accounting concept demands ecosystem classifications that are suitable for statistical analysis and accounting and that also allow for the results to be compared across countries.

As fully explained in Annex A, for the pilot tests of the NCAVES-Mexico project the delineation of Mexico's ecosystems was based on the intersection of three criteria:

- <u>Vegetation type or land-use class</u>. The vegetation classes provide the basic structure for delineating/distinguishing ecosystem types and for examining their extent and their changes over time. They also provide a link to the production of ecosystem services and to their users. The vegetation information used in these studies was derived from INEGI's vegetation and land-use charts scale 1:250,000. In INEGI's charts native vegetation is classified according to INEGI's own vegetation and land-use classification system, in which vegetation types are described in terms of a combination of floristic, life-form and ecological characteristics, and their relationship to environmental conditions. INEGI's full classification distinguishes 59 different types of natural vegetation that occur in Mexico. For ecosystem accounting purposes, we used a simplified version of INEGI's classification system in which vegetation types are aggregated into larger groups for easier mapping and correlation with other data. This simplified classification scheme encompasses 14 vegetation types (distinguishing well-preserved or primary vegetation from degraded or second-growth vegetation), five land-use and two ancillary classes. Fig.2a below shows the geographical distribution of the different types of vegetation or land-use in Mexico.
- <u>Land tenure</u>. Land tenure refers to the ownership of land and the purpose of its management. The following types of land tenure occur in Mexico: communally-owned land, encompassing *ejido* and agrarian/indigenous communities (see Fig.2b below), and other property regimes (including small landholdings, national lands and others). Communally-owned land (or social property) encompasses approximately 102 million ha, or over 52% of the country's territory. Social property in Mexico implies —among other features— that forests, shrubland, and other natural vegetation (*tierras de uso común*, land for collective use) are owned and managed by the entire community or *ejido*, rather than by individuals or households. Land tenure was determined based on the cadastre kept by the National Agrarian Registry. This data set denotes areas of ownership as land parcels defined by polygons.
- Presence of (federal) <u>protected areas</u> (see Fig.2c below). As of 2019, 182 federal protected areas have been decreed in Mexico, with different levels of protection ranging from sanctuaries to biosphere reserves. These encompass over 90 million hectares, including 21 million hectares in terrestrial areas and 69 million hectares in marine areas, amounting to 10.7% of Mexico's total terrestrial area and 22% of its territorial sea, respectively (compare with the 17% of terrestrial area and 10% of coastal and marine area aimed to under Aichi's Target 11). As is the case in other Latin American countries, most protected areas in Mexico are inhabited by rural populations that carry out subsistence/productive activities (farming, cattle-ranching, forestry, etc.) but under the restrictions



imposed by the protected area regulations.

The data sources and methods used to produce the spatially explicit delineation and representation (see example in Fig. 2d for 2014) of the ecosystems thus defined are described as Procedimiento 1 in Annex B.





Fig.2a. INEGI's Series 6 chart portraying the extent and distribution of Mexico's vegetation and land-use types as of 2014. INEGI's Vegetation and land-use types are aggregated as *per* the simplified classification used by CONAFOR for the greenhouse gas emissions inventory of the Mexican LULUCF sector.



Communal land tenure



Fig.2b. Distribution of Mexico's communal lands, including indigenous communities' and *ejido* lands, as per the National Agrarian Registry.



Fig.2c. Distribution of Mexico's federal Protected Areas as of 2014.





Fig.2d. Distribution of Mexico's ecosystems as of 2014, as defined for the NCAVES-Mexico project. See details in the text.



6. Ecosystem extent

The next step in the SEEA-EEA workflow is estimating the ecosystems' extent at different points in time. As fully explained in Annex A, INEGI's Vegetation and Land Use, scale 1:250 000, charts were used as the basis for estimating the extent of Mexico's ecosystems and its changes over time (as of 2002, 2007, 2011 and 2014), using the methods described as Procedimiento 2 in Annex *B*. As per the *Technical recommendations in support of the SEEA-EEA* (UN, 2017), results are to be summarized in extension account tables showing the opening and closing values for the study periods.

Fig.3a below shows the extent of the country's ecosystems as of 2014. The major ecosystem types include xerophytic shrublands and temperate and tropical forests, which together cover some 62.8% of the country's territory. Agricultural lands devoted to annual crops cover almost 16% of the territory. While a significant fraction (> 12%) of major ecosystems such as xerophytic shrublands, evergreen tropical forests and mountain cloud forest is included in federal protected areas (see Fig.3b below), only a small fraction of other major ecosystems —such as deciduous and semideciduous tropical forests— enjoys this sort of protection. Nevertheless, over 40% of the remaining extent of key ecosystems such as mangrove forests (included in woody hydrophytic vegetation) are currently included in protected areas.

Table 2 below summarizes the changes in the extent of the country's ecosystems over time. As can be seen, the extent of the major natural ecosystems has undergone considerable reductions. From 2002 to 2014, the total extent of the country's semideciduous tropical forests experienced a net reduction of 776 thousand hectares (a 16.2% loss); that of woody xerophytic shrublands was reduced in 588 thousand hectares (2.8% loss); evergreen tropical forests in 581 thousand hectares (a 5.5% loss); non-woody xerophytic shrublands in 381 thousand hectares (1% loss); deciduous tropical forests in 171 thousand hectares (0.9%); and coniferous forests in 91 thousand hectares (0.5% loss). Such reductions in the extent of natural ecosystems have been evidently due to their conversion to agricultural land (devoted particularly to annual crops) and to urban areas and other human settlements. From 2002 to 2014, the total area devoted to annual crops agriculture increased in 1 774 thousand hectares (a 5.9% increase).

As can be seen in Fig. A.1 (in the <u>Appendix</u>), much of these conversions has taken place mainly in ejido lands, whereas only remarkably minor reductions have occurred in indigenous community lands; this shows the importance of land tenure on decisions about the management and use of land and natural resources. The protection that protected areas provide to natural ecosystems, preventing their conversion to other land uses, has also extremely important. In net terms, all the above-mentioned reductions in the extent of natural ecosystems took place in non-protected zones, whereas their extent within protected areas remained stable or increased over the study period. From 2002 to 2014, the extent of semideciduous tropical forests within protected areas increased in 27 thousand hectares (a 10.5% increase); that of woody xerophytic shrublands in 119 thousand hectares (3.2% increase); evergreen tropical forests in 77 thousand hectares (a 5% increase); non-woody xerophytic shrublands in 490 thousand hectares (11.9% increase); deciduous tropical forests in 276 thousand hectares (27.2% increase); and coniferous forests in 106 thousand hectares (5.3% increase) and. This is mostly a result of the steady expansion of the national system of protected areas over the last 30 yr, and particularly rapidly during the last 15yr. In fact, the total extent of mangrove ecosystems (included in the Woody hydrophytic vegetation class) has increased over time (42 thousand hectares from 2002 to 2014), thanks to the designation of key coastal zones as protected areas: The extent of protected woody hydrophytic vegetation increased in 81 thousand hectares over the same period of time.











Figure 3b. Fraction of Mexico's natural ecosystems included in federal protected areas, as of 2014.

	Ecosystem type										
	Coniferous forest	Oak woodland	Mountain cloud forest	Evergreen tropical forest	Semideciduous tropical forest	Deciduous tropical forest	Woody xerophytic shrubland	Non-woody xerophytic shrubland	Grassland		
2002	16,950,421	15,673,454	1,854,718	10,540,847	4,778,123	18,036,039	21,201,247	37,113,590	31,620,377		
Change (%)	-0.22	-0.04	0.92	-2.27	-6.69	-1.57	-1.01	-0.74	-1.47		
2007	16,912,441	15,667,720	1,871,816	10,301,077	4,458,698	17,752,808	20,986,194	36,837,334	31,156,391		
Change (%)	-0.26	1.35	-1.71	-1.38	-8.88	1.07	-1.52	-0.43	-1.32		
2011	16,867,640	15,878,664	1,839,732	10,158,886	4,062,693	17,943,409	20,666,943	36,677,837	30,745,606		
Change (%)	-0.05	-0.04	-0.81	-1.96	-1.49	-0.44	-0.26	0.15	0.45		
2014	16,859,370	15,872,893	1,824,809	9,959,423	4,002,176	17,865,052	20,613,691	36,733,052	30,885,225		

	Table 2. Ecosys	stem extension	account table.
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	Ecosystem type									
	Other non-									
	Other woody	woody	Woody	Non-woody						
	vegetation	vegetation	hydrophytic	hydrophytic			Perennial	Human	Forest	Water
	types	types	vegetation	vegetation	Other lands	Annual crops	crops	settlements	plantations	bodies
2002	428,423	159,937	981,702	1,140,618	1,480,616	29,421,034	1,656,757	1,267,254	32,114	1,193,431
Change (%)	-0.08	-1.04	3.02	3.16	-2.95	4.48	3.61	26.82	16.32	4.42
2007	428,060	158,272	1,011,357	1,176,668	1,437,003	30,738,869	1,716,581	1,607,122	37,354	1,246,151
Change (%)	-2.12	-0.14	3.37	0.68	2.22	0.98	4.78	33.40	74.70	117.40
2011	418,974	158,047	1,045,404	1,184,649	1,468,875	31,041,495	1,798,681	2,143,929	65,257	2,709,128
Change (%)	-0.41	-0.25	0.95	-0.15	-0.82	0.40	3.39	1.86	15.08	-0.42
2014	417,251	157,652	1,055,376	1,182,873	1,456,795	31,165,524	1,859,572	2,183,804	75,100	2,697,754



7. Ecosystem condition

The third step in the SEEA-EEA framework workflow (see Fig.1 above) consists in compiling information that describes the ecosystems' condition. This allows examining how ecosystems in the study area change over time and how those changes may affect their ability to supply ecosystem services.

As of today, the measurement of ecosystem condition is still a matter of debate (Rendon *et al.*, 2019) and no single —or entirely appropriate— measure or indicator of ecosystem condition exists. Different variables or indicators thought to be relevant for the ecosystems present in the study area may be used instead. As fully discussed in <u>Annex A</u>, due to the scarcity of data with the required features and availability, only two variables/indices could be used for mapping and assessing the condition of Mexico's ecosystems and its changes over time:

- 1.- Conservation status of vegetation, and
- 2.- an index aimed to denote the impact of anthropogenic activities on natural ecosystems.

7.1.- Conservation status of vegetation

The INEGI's Vegetation and Land Use charts used to estimate the extent of Mexico's ecosystems at their changes over time (see section **6. Ecosystem extent** above) also distinguish the successional stage or conservation status (primary or relatively well preserved *vs.* second-growth or degraded) of each vegetation type, recognizing three different phases (or successional stages) of second-growth vegetation:

- Tree-dominated second-growth vegetation,
- Shrub-dominated second-growth vegetation, and
- Predominantly herbaceous second-growth vegetation

Hence, from the information contained in INEGI's Vegetation and Land Use charts, the extent of the country's natural terrestrial ecosystems with different conservation status and its changes over time can be readily examined —both in tabular and in a spatially explicit manner. This actually produces two interlinked variables: Extent of primary (or well-preserved) vegetation and extent of second-growth (or degraded) vegetation. According to the latest findings from the SEEA-EEA revision process (Keith *et al.*, 2019; Czúcz *et al.*, 2019), a reference level would be necessary to turn these two variables into a proper indicator(s) of ecosystem condition. However, as discussed in <u>Annex A</u>, there is no practicable manner to set a suitable, scientifically-supported reference level for these variables. It can be argued, however, whether this is at all necessary, as what really matters is the observed change trend that these variables follow over time: An increase in the extent of well-preserved (accompanied by a concomitant reduction in the extent of degraded vegetation) would clearly denote an improvement in the ecosystems' condition, and *viceversa*.

The data and procedures used to construct the ecosystem condition accounts based on the two variables denoting the conservation status of vegetation are fully described as Procedimiento 3.1 in Annex *B*. The results obtained are summarized in ecosystem condition account tables showing the opening and closing values for the study periods. Figure 4 below summarizes these results in a graphical manner. As can be seen, not only has the extent of most natural ecosystems of the country been reduced over time but, in many cases, this has also been accompanied by a degradation process by which former primary forests have been replaced by disturbed or early successional forests. Between 2002 and 2014, the extent of degraded non-woody xerophytic shrublands increased in 553 thousand hectares, while that of well-preserved shrubland decreased in 944 thousand hectares; degraded deciduous tropical forests increased in 404 thousand hectares, while well-preserved forests were reduced in 575 thousand hectares; and degraded coniferous forests increased in 331 thousand hectares, while well-preserved ones decreased in 422 thousand hectares. In fact, the full extent of this degradation process might be not be



entirely apparent from this analysis, as the simplified classification used for this pilot studies lumps together truly primary forests with tree-dominated, second-growth forests. By contrast, in evergreen and semideciduous tropical forests and woody xerophytic shrublands, conversion to other land uses has predominated over degradation processes, leading to an overall reduction in their total extent. The protection afforded by protected areas is also clearly evident as most of the above-mentioned increases in the extent of degraded natural ecosystems took place in non-protected zones.



Fig.4 Changes over time in the conservation status (primary and well-preserved vs. degraded or second-growth) of vegetation in Mexico's terrestrial ecosystems, as estimated from INEGI's Vegetation and land-use charts for 2002, 2007, 2011 and 2014. Note the different scale used for non-protected (left vertical axis) and protected areas (right vertical axis).



7.2.- Human footprint index

The Human footprint index aims to denote, in relative terms, the extent to which natural environments have been modified or transformed by human activities. The index is computed by estimating/assessing —in a spatially explicit manner— both, the extent and intensity of the transformation caused by various human activities (for which spatially explicit information is available) including urbanization and human settlements; agriculture and aquaculture; roads and other infrastructure; industry, mining, etc.

As such, the Human footprint index does not actually constitute an indicator of the ecosystems' condition; rather, it summarizes a range of socio-economic pressures and drivers of change that affect the ecosystems' condition. According to the recent recommendations from the SEEA-EEA revision process (Keith *et al.*, 2019; Czúcz *et al.*, 2019), the use of pressure indicators in ecosystem condition accounts should be avoided. Instead, the underlying ecosystem characteristics/properties being affected by such pressures should be identified and measured and used as condition indicator(s). However, as I have argued in Annex A, the affected ecosystem characteristics might be difficult to identify and/or suitable information about them might not be available. Given the strong causal relation existing between pressures and ecosystem condition, in cases where direct indicators of ecosystem condition. Thus, the possibility of using the Human Footprint as an indicator of the socio-economic pressures affecting Mexico's ecosystems was explored.

The Human Footprint was recently evaluated by SEMARNAT for the entire country with data for 2011 and was included in the 2015 edition of Mexico's *State of the Environment Report*. In response to an *ad hoc* request, SEMARNAT decided to produce an updated version of the index with data for 2014-2015. The data sources and methods used to construct the Human Footprint Index are fully described as Procedimiento 3.2 in Annex B. The results obtained are summarized in form of condition account tables showing the opening and closing values for the study periods; these tables are included in the Appendix.

Fig. 5 shows the geographic distribution of the Human Footprint values across the country for 2011 (Fig.5a) and 2014-2015 (Fig.5b). Although the maps show a significant increase in the extent of highly and very-highly transformed areas from 2011 to 2014/2015, part of this change might be actually due to differences in the data sets used for constructing these maps, particularly the mining concession data (see explanation in Annex B; SEMARNAT and INEGI have begun collaboration to address this methodological issue and produce an improved version of the Human Footprint Index for Mexico). Nevertheless, beyond this artifact, it is useful to examine where these changes have occurred. Fig. 5c below shows the geographic distribution of those areas where HF values changed significantly from 2011 to 2014/15 either in the positive (*i.e.*, their degree of transformation decreased; shown in shades of green in Fig.5c) or —more importantly— the negative direction (*i.e.*, the extent and/or intensity of the transformation caused by various human activities increased; shown in shades of red in Fig.5c). Many of the negative changes in the northern half of the country, particularly those across the Mexican Plateau, seem to be due to the more detailed mining concession data used for constructing the 2014/2015 map. However, the negative changes in the Yucatan peninsula and the States of Chiapas and Oaxaca are due to the expansion of agricultural and other economic activities therein.

Figure 6 below summarizes, in a graphical manner, the changes in the relative degree of transformation caused by various human activities on the country's ecosystems over time. As can be seen (Fig.6a), although a significant fraction (over 80%) of the country's natural ecosystems still appears as only scarcely or slightly affected/transformed by human activities, all the ecosystems have been affected, to a lesser or larger degree, in different parts of their distribution range. The degree of transformation has also noticeably increased over the period 2011-2014/15 in all ecosystems. As expected (see Fig.6b),



ecosystem assets within protected areas have been noticeably less transformed by human activities than their counterparts in non-protected zones. Similarly, the protection afforded by their designation has prevented the degree of transformation to increase noticeably over time.



Fig.5. Charts showing the geographic distribution of Human Footprint values in Mexico as of a) 2011 (top panel) and b) 2014-2015 (bottom panel). See details in the text.



Changes in Human Footprint 2001 -> 2014/15



Fig.5c. Geographic distribution of changes (increase/reduction) in Human Footprint values from 2011 to 2014-2015. See details in the text.





Fig.6 Changes over time (2011 to 2014/2015) in the impact of human activities on the main ecosystems in Mexico, as assessed by the Human Footprint Index. 6a (upper panel): Overall ecosystem extent with different levels of transformation by human activities; 6b (bottom panel): Data disaggregated by protection status (PA, protected areas).

8. Mapping and valuing ecosystem services in Mexico

In the previous sections, the country's ecosystem assets were identified and delineated in a spatially explicit manner, and their extent and condition were evaluated. The following steps in the SEEA-EEA framework consist in measuring —first, in physical terms— the supply and use of the services the ecosystems provide, and, finally, valuing these in monetary terms (see Fig.1 above; UN, 2017). A distinctive feature of the SEEA-EEA framework is its explicit ambition for linking ecosystems with economic production and consumption and thus integrating ecosystem services into the national economic accounts (UN, 2017). In order to achieve this, the SEEA-EEA framework imposes some very strict conceptual and methodological requirements, as follows:

First, in this framework, ecosystem services are narrowly defined as the <u>contributions</u> that ecosystems make to the production of benefits used in economic or other human activities. This definition emphasizes the fact that, in many cases, the benefit of ecosystem services is realized only after economic agents (*e.g.* ecosystem managers, farmers, industry, etc.) have invested inputs (*e.g.*, capital and labour to manage the ecosystem, harvest the services, etc.) to transform the flows from ecosystems into



the goods and services (products) that are actually used by and benefit people. In the SEEA-EEA framework this is a crucial distinction to make, particularly at the time of evaluating ecosystem services in physical terms when, for a given good or service derived from ecosystems, the actual contribution of the ecosystem should be disentangled from what is generated by/through human inputs and interventions. Making this distinction is also important when valuing the ecosystem services in monetary terms as most of such human inputs and interventions (*e.g.*, capital and labour) are part of the economic process and are already included in national economic accounts —and should not be counted twice.

Secondly, the SEEA-EEA framework focuses only on provisioning, regulating, and cultural ecosystem services (UN, 2017). These are regarded as final ecosystem services that directly support/contribute to economic and human activity and thus constitute a direct link between ecosystems and the economy. Supporting ecosystem services such as nutrient cycling, soil formation, etc. are considered as intermediate services and while their importance for understanding relationships and dependencies within and between ecosystems is recognized, they are excluded from the SEEA-EEA accounts. To avoid ambiguities in the identification of ecosystem services, a unified (or, at least, a widely-used) classification of ecosystem services (e.g., CICES, 2018) should be used.

Thirdly, each ecosystem generates a set of various ecosystem services. However, under the SEEA-EEA framework, benefits from such services are only realized —and enter the economy— when they are actually used (and benefitted from) by economic units (government, business, household, individual person). Two important implications of this are that only those services for which a corresponding user and a benefit (in terms of human well-being) can be identified are encompassed by the SEEA-EEA accounts and, for such services, only the fraction of the total flow that is actually used is included in the accounts (see below). For products (such as food, timber, potable water, etc.) already included in standard economic accounts, users and beneficiaries are well identified; however, the beneficiaries of ecosystem services such as climate regulation, clean air, etc. have first to be identified in each particular instance. For example, the benefit of services such as the regulation of chemical composition of atmosphere only materialises if some people are actually affected by such process.

Fourthly, the SEEA-EEA framework distinguishes between ecosystem service capacity (or supply) and flow (also referred to as provision or demand) (UN, 2017). Capacity is the theoretical or potential capacity of the ecosystem to supply a given ecosystem service, whereas flow represents the actual provision and use of the service, which is a function of the users' demand. Following from the previous considerations, it is the ecosystem service provision what is included in the SEEA-EEA accounts. However, both components are important as they help to assess how sustainably the ecosystem service is being used: If the demand for (and provision of) a given ecosystem service surpasses the ecosystem's capacity to supply it, the use is unsustainable in the long-term and would lead to ecosystem degradation.

Finally, if the value of ecosystem services is to be meaningfully and directly integrated with standard national economic accounts, then their valuation in monetary terms should use the same value concept used in national accounts, *viz.*, the exchange value (UN, 2014b). The exchange value of an ecosystem service is the value that reflects the price at which the service would be exchanged between the users of the service (buyers) and the relevant ecosystem (producer or seller) if an observable market for it existed —but excluding any profit or consumer surplus. Since for many ecosystem services no observable market exists, then non-market valuation techniques have to be used to estimate an appropriate monetary value. However, as discussed in the *Country Assessment on Natural Capital Accounting and Valuation of Ecosystem Services in Mexico*, only a few of the commonly used non-market valuation techniques yield monetary values that are appropriate for SEEA-EEA accounting purposes.



The above described requirements and restrictions were borne in mind when designing the methodological and analytical approaches to be used for pilot testing the SEEA-EEA framework in Mexico. A specific spatial model was developed for each of the four ecosystem services chosen (viz., carbon sequestration, surface water supply, food crop production, and coastal protection by mangrove ecosystems). In most cases, the models were developed by using INEGI's vegetation charts as a framework for spatializing statistical and administrative records data. The four latest editions of INEGI's vegetation charts (corresponding approximately to the years 2002, 2007, 2011 and 2014) were used for modelling the ecosystem services —unless indicated otherwise in the model descriptions in those cases where data were insufficient or not available. The modelling outcomes were then used to construct ecosystem accounting tables, for both, the country-wide and State-level. The data sources and methods used to construct the models and analyse the results are fully described in Annex B. In the following sections I briefly outline the modelling approach used and describe in detail the results obtained in each pilot study.

8.1 Carbon storage and sequestration

The pilot study on carbon storage and sequestration by Mexico's ecosystems was carried out with data and technical support from the Technical Unit Specialized in Monitoring, Reporting and Verification (UTE, for its acronym in Spanish) of the Forestry Monitoring Office of CONAFOR. I used some of the very same data and adapted some of the methods that UTE utilizes for estimating greenhouse gas emissions from the land use, land-use change and forestry sector (LULUCF) in Mexico, as part of the National Greenhouse Gas Emissions Inventory (SEMARNAT-INECC, 2018). The data sources, data processing and estimation methods used by UTE for this purpose are fully described elsewhere (*Strengthening REDD + and South-South Cooperation*, 2015); all of them closely follow the Intergovernmental Panel on Climate Change's guidelines.

The two-stage process used by UTE to estimate greenhouse gas emissions from the LULUCF sector consists, first, of using field data from the National Forest Inventory (compiled by CONAFOR over 5-yr cycles) to estimate the average amount of carbon stored (in tm of C/ha, or carbon density) in each ecosystem type. Since two full sampling cycles of the National Forest Inventory (NFI) have been completed to date, C density estimates are available for two points in time, namely, 2004-2009 and 2009-2014. Then, the difference in carbon stocks between the two sampling dates is used to estimate the rate of carbon sequestration/loss (tm of C/ha/yr) for each ecosystem type. The second stage consists of using INEGI's Vegetation and Land-use charts (for the dates most closely matching the forest inventory sampling cycles) to extrapolate the carbon density values and carbon sequestration/loss rates to obtain country-wide estimates, for each ecosystem type, of carbon stocks (for two points in time) and the amount of carbon sequestered (or lost) over the study period.

The country-wide averages estimated by CONAFOR subsume any geographical variability in the country's ecosystems and in their C stock and sequestration values due to environmental, compositional or structural variations. Thus, they would not be suitable for the NCAVES-Mexico project, as the SEEA-EEA framework demands spatially explicit estimates of ecosystem services supply. In order to obtain regionalized estimates of carbon density and sequestration rates, I used the map of North American Terrestrial Ecoregions (level II) to first stratify the NFI field data by the 15 level II ecoregions recognized for Mexico. Ecoregions do provide an integrated, ecologically-based, spatially-explicit set of criteria for stratifying the NFI data for this purpose: Ecoregions are areas of general similarity in ecosystems and in the type, quality, and quantity of environmental resources. In the map, ecoregions are delineated based on the analysis of patterns and composition of both biotic and abiotic features such as geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology, that affect or reflect differences in ecosystem quality and integrity. Level I ecoregions divide North America into 15 broad ecological



regions that highlight major ecological areas in the continent. The 50 level II ecological regions provide a more detailed description of the large ecological areas nested within the level I regions and are useful for national and sub-continental overviews of ecological patterns (<u>CEC, 1997</u>).

Then, I replicated the same methods used by UTE to first obtain estimates —for each ecosystem type within each ecoregion— of C density for each of the two sampling periods. These estimates included C stored in the above- and below-ground biomass but not in the soils' organic fraction (as discussed in Annex A, data available for soil organic C lack the spatial and temporal consistency necessary for these purposes). Then, rates of C sequestration/loss for each ecosystem type within each ecoregion were estimated from the difference in C stocks between the two sampling dates. The C flows thus estimated encompass C sequestered by (or lost from) vegetation due to growth of trees, recruitment of new trees, emissions due to wildfires, collapse of dead standing trees, and losses due to logging, but not emissions from soil. A model to spatially display the variations in C density and C sequestration/loss across the country was constructed by combining the regional-level estimates with series 4 and 5 of INEGI's Vegetation and Land Use charts (corresponding to 2007 and 2011, respectively), and the Terrestrial Ecoregions chart. The data sources and procedures used for these purposes are fully described as Procedimiento 4.1 in Annex B. This procedure ensures that the estimates of C sequestration rates and C stocks obtained are fully consistent with those officially reported by CONAFOR to be included in the National GHG emissions Inventory for the LULUCF sector that is submitted by the Mexican government to the UNFCCC in its national communications.

<u>Carbon stocks in Mexico's terrestrial ecosystems</u>.- Fig. 7 shows the geographic variations in C stocks (in terms of the average amount of C stored per ha) in Mexican ecosystems for two points in time: 2004-2009 (Fig.7a) and 2009-2014 (Fig.7b); these dates correspond to the two sampling cycles of the National Forest Inventory that have been completed so far. As can be seen, the humid tropical forests of the South and Eastern part of the Yucatán peninsula are the ecosystems containing the highest C densities; these are followed by the temperate forests of the Western Sierra Madre and the Trans-Mexican Volcanic Belt. It is also noticeable the overall increase in C density between the two sampling periods, likely due to the abundance of actively growing secondary forests in Mexico (see below).

Table 2 shows the country-wide estimates of C density and C stocks in Mexican ecosystems for 2004-2009 and 2009-2014. The total amount of C stored in each ecosystem type is the product of its extent and C density. Thus, the country's largest stocks of C are stored in the over 16 million ha covered by coniferous forests, followed by those in the remaining 10 million ha of evergreen tropical forests, and the nearly 16 million ha covered by oak forests and woodlands. Although the highest C densities are found in the well-preserved mountain cloud forests of the Western Sierra Madre and the fast-growing commercial forest plantations of Southern Mexico, these ecosystems cover only very small areas. The total amount of C stored in the country's ecosystems was estimated to be 1 854.8 Mtm C as of *ca*. 2007and this increased to 2 103.5 Mtm C by *ca*. 2011 (see Table 2).

<u>Carbon sequestration by Mexico's terrestrial ecosystems</u>.- The SEEA-EEA framework does not consider carbon storage as an ecosystem service (UN, 2017) as this does not constitute, *per se*, a flow from ecosystems to society and is, therefore, not included in ecosystem accounts. The relevant ecosystem service is carbon sequestration, that is, the removal of carbon dioxide from the atmosphere and its storage in biomass and soils in ecosystems. Carbon sequestration is related to Net Ecosystem Productivity, that is, the difference between Gross Primary Productivity and Total Ecosystem Respiration (including autotrophic respiration from plants and heterotrophic respiration from soil).

Carbon sequestration (or loss) can be indirectly measured based on the net change in carbon stocks in ecosystems, as this is the balance between additions due to plant growth and reproduction and



losses due to combustion, decomposition and removal of biomass from the ecosystem. A negative net change in carbon stocks implies a loss of carbon from ecosystems and a contribution to the country's greenhouse gas emissions. A positive net change in carbon stocks represents the ecosystem service of carbon sequestration.

Table 2 shows that most ecosystems increased their C stocks between the two sampling periods, indicating that they actively sequester carbon from the atmosphere. Natural ecosystems with the highest rates of C sequestration are the well-preserved mountain cloud forests and semideciduous tropical forests. However, the highest rates of C sequestration occur in the fast-growing commercial forest plantations of Southern Mexico. The total contribution of each ecosystem type to the sequestration of atmospheric carbon dioxide is the product of its extent and its C sequestration rate. Considering the ecosystems' extent as of 2007, evergreen tropical forests and coniferous forests are the ecosystems that most contribute to this ecosystem service, followed by deciduous tropical forests, oak woodlands and forests and semideciduous tropical forests (Table 2). Together, natural grasslands and pasturelands also sequester an important amount of atmospheric carbon dioxide each year.

Fig. 8 shows the geographic variations in the average amount of C that would be yearly sequestered/lost by ecosystems present in Mexico in 2007, were they to remain undisturbed (*i.e.*, with no change in extent or condition). Although some ecosystems act as C sources (negative rates), most of them sequester atmospheric CO_2 at moderate rates (< 2.5 tm C/ha/yr); there are a few particular cases (such as the tropical forests in Los Tuxtlas region in the Gulf of Mexico coast) that show very high rates (*ca.* 18 tm C/ha/yr) of C sequestration. These very high values may be due, at least in part, to sampling error; data from the third sampling cycle (due by 2020) of the forest inventory should help confirm/correct these.

<u>Relationship between C sequestration and ecosystem condition</u>.- As expected, in all cases, second-growth or degraded ecosystems contain less biomass per unit area and, thus, a smaller C stock than their primary or well-preserved counterparts. However, in most ecosystem types, the C sequestration rate of second-growth or degraded ecosystems is proportionally higher (considering their lower biomass density) than that of primary or well-preserved ones. This difference is likely due to second-growth forests being in a rapid growth stage. Thus, the contribution of second-growth or degraded ecosystems to the ecosystem service of C sequestration cannot be neglected.

I was unable to examine the relationship between C sequestration and the degree of transformation of natural ecosystems by human activities (as assessed through the Human Footprint Index). The method used assigns the same C density value to all areas covered by the same vegetation type within a given ecoregion, regardless of their degree of transformation.



			2004-2009		200	9-2014		
		Vegetation conservation	Mean C stock (tm C/ha)	Total C stock in biomass (tm C)	Mean C stock (tm C/ha)	Total C stock in biomass (tm C)	Mean C sequestration rate (tm C/ha/yr)	Total C sequestration (tm C/yr)
	Coniferus forest	Primary	38.4	496,155,981	40.7	526,088,019	0.53	7,446,345
	connerus rorest	Second growth	24.2	92,172,037	27.1	104,402,487	0.43	1,786,754
	Oak woodland	Primary	23.3	247,048,636	25.2	269,613,726	0.33	3,555,156
		Second growth	16.0	76,026,378	18.4	89,998,751	0.39	1,766,827
	Mountain cloud	Primary	45.1	57,788,046	55.7	71,223,753	1.43	1,909,159
	forest	Second growth	20.6	11,182,477	26.0	13,026,322	0.78	359,883
	Evergreen	Primary	39.0	306,330,967	43.6	347,875,329	0.80	8,793,739
	tropical forest	Second growth	18.0	37,607,213	19.3	37,603,863	0.34	625,616
	Semideciduous	Primary	26.8	80,283,840	32.5	92,384,660	1.25	3,683,920
	tropical forest	Second growth	14.3	20,929,817	18.3	22,222,631	0.75	1,240,476
	Deciduous	Primary	13.5	146,642,070	15.1	161,863,526	0.36	3,681,826
	tropical forest	Second growth	9.1	63,150,913	11.5	82,503,170	0.43	2,898,743
	Woody	Primary	2.4	43,459,427	2.6	47,492,495	0.0	921,651
Ecosystems	shrubland	Second growth	1.8	4,352,804	2.0	4,822,620	0.03	68,730
	Non-woody	Primary	0.4	12,278,627	0.4	12,402,913	0.008	259,938
	shrubland	Second growth	0.5	1,045,531	0.3	501,330	- 0.04	- 100,410
	Other woody	Primary	9.7	1,163,863	10.7	1,112,380	0.13	10,951
	vegetation types	Second growth	6.8	1,096,659	8.1	1,242,629	0.09	- 4,679
	Other non-	Primary	3.2	685,186	2.1	407,919	- 0.17	- 39,499
	woody	Second growth	ND	ND	ND	ND	ND	ND
	Woody	Primary	16.0	14 190 146	19.3	18 045 283	0.80	755 201
	hydrophytic	i i i i i i i i i i i i i i i i i i i	10.0	1,150,110	15.5	10,013,203	0.00	733,001
	vegetation	Second growth	9.0	548,152	15.4	1,111,170	1.26	100,605
	Non-woody	Primary	2.3	2,758,811	3.0	3,483,945	0.01	25,805
	nyaropnytic	Second growth	ND	ND	ND	ND	ND	ND
	Grassland	NA	6.3	136.255.018	8.5	190,268,788	0.24	8,439,289
	Forest		44.6	1 688 869	50.0	3 778 558	1 05	72 251
	plantation	NA	44.0	1,000,009	59.9	3,770,558	1.95	72,231
		Cour	ntry-wide total	1,854,841,468		2,103,476,269		48,258,880

Table 2. Carbon storage (tm) and C sequestration rates (tm C/yr) by Mexican ecosystems. See explanation in the text.





Fig.7. Carbon density (tm C/ha) in Mexican ecosystems as of 2004-2009 (7a: top panel) and 2009-2014 (7b: bottom panel). See explanation in the text.







Fig.8. Potential C sequestration (tm C/ha/yr) by Mexican ecosystems. See explanation in the text.



<u>Valuing the C sequestration services provided by Mexico's ecosystems</u>.- A prerequisite for valuing ecosystem services is identifying the beneficiaries and the actual use that they make of the service. As carbon emissions at a global level are far larger than what ecosystems can sequester, all carbon sequestered by ecosystems constitutes a flow to society. Therefore, for the ecosystem service of C sequestration, the service supply equals its provision or flow. In addition, its contribution to climate change mitigation benefits people in the study area or even beyond —even globally.

<u>Carbon offsets</u>.- Clearly, the exchange value of carbon sequestration by ecosystems cannot be directly determined. According to the SEEA-EEA framework (UN, 2014), for ecosystem services for which market prices are not observable, one possible alternate approach would be to base their valuation on market price equivalents, *i.e.*, the market price of similar goods or services.

As discussed in the Country Assessment on Natural Capital Accounting and Valuation of Ecosystem Services in Mexico, since the early 1990s market-based mechanisms have been designed and implemented to either off-set negative environmental impacts (e.g., Emission Trading Systems) or to monetize the provision of environmental services (Payment for Environmental Services schemes). Thus, prices used in such market-based schemes could be used as the basis for valuing the ecosystem service of C sequestration by Mexican ecosystems. Although Mexico's Emission Trading System has not yet been implemented, it can be argued that this ecosystem service has a benefit for climate change mitigation both nationally and internationally. Thus, the value of carbon sequestered by Mexico's ecosystems can be equated to the price that could be paid in other (voluntary or compliance) carbon markets to offset GHG emissions made somewhere else. In 2016, forestry and land-use carbon offsets in voluntary markets in Latin America commanded an average price of US\$4/tCO₂e (Forest Trends, 2017). Using this price, the total value of the C sequestration service potentially provided by Mexico's ecosystems existent in 2007 (were they to remain undisturbed, *i.e.*, not subject to degradation nor deforestation) would be US\$708 million per vr (see Table 3). On a per unit area basis, the most "valuable" ecosystems are the wellpreserved mountain cloud forests and semideciduous tropical forests, along with the fast-growing commercial forest plantations of Southern Mexico. However, taking into account the ecosystems' extent as of 2007, the areas covered by evergreen tropical forests and coniferous forests are the ones that most contribute to the total value of this ecosystem service, followed by deciduous tropical forests, oak woodlands and forests and semideciduous tropical forests (Table 3).

<u>Social cost of carbon</u>.- The high number and diversity of carbon markets currently in operation (Forest Trends, 2017), and the fact that the their prices reflect, to a large extent, the supply-demand dynamics and/or the regulatory framework existing in the different countries (UN, 2014), might make the price of C offsets not be the best approach to value the ecosystem service of C sequestration. An alternate, perhaps better, approach might be valuing the carbon sequestration service in terms of the cost of the damage caused to society by the emission of an additional tm of carbon (IAWG, 2013).

The marginal social cost of carbon (SCC)_represents the economic cost associated with the climate damage (or benefit) resulting from the emission (or sequestration) of an additional tm of CO₂. Numerous estimates of SCC have been produced over the last 30 yr, based on widely different assumptions about the social discount rate, economic growth and climate sensitivity. Recent estimates of SCC range from approximately US\$10 to as much as US\$1,000 per tm of CO₂ —clearly, many challenges and opportunities exist for improving the SCC estimates (National Academies of Sciences, Engineering and Medicine, 2017). Perhaps the most widely used, global estimates of SCC are those produced by the US Environmental Protection Agency. EPA's latest estimates are US\$12, US\$42 and US\$62 per tm of CO₂ emitted in 2020 for 5, 3 and 2.5% discount rates, respectively (IAWG, 2013). However, as global estimates of SCC subsume any geographic differences in climate damages, climate and socio-economic conditions, and in the country's contributions to the global SCC, Ricke *et al.* (2018) recently estimated country-level SCCs. The



estimated_median of the Social Cost of Carbon for Mexico for a reference scenario in 2020 is US32.5/tm of CO₂e.

Using this value, the C sequestration service potentially provided by Mexico's ecosystems existent in 2007 (were they to remain undisturbed, *i.e.*, not subject to degradation nor deforestation) would then be valued in as much as US\$5.8 billion per yr (see Table 3).

		Vegetation conservation	Total C sequestration (tm C/yr)	Total CO ₂ e sequestration (tm CO ₂ e/yr)	Potential value if traded as offsets (US\$/yr)	Value in terms of social cost of C(US\$/yr)
	Coniforus forost	Primary	7,446,345.2	27,306,492	109,225,969	887,461,002
	connerus iorest	Second growth	1,786,753.9	6,552,205	26,208,821	212,946,673
	Oakwooldand	Primary	3,555,156.4	13,037,114	52,148,457	423,706,210
	Oak wooldallu	Second growth	1,766,827.0	6,479,131	25,916,526	210,571,771
	Mountain cloud	Primary	1,909,159.2	7,001,078	28,004,310	227,535,021
	forest	Second growth	359,882.6	1,319,725	5,278,902	42,891,078
	Evergreen tropical	Primary	8,793,739.0	32,247,520	128,990,081	1,048,044,412
	forest	Second growth	625,616.4	2,294,198	9,176,791	74,561,430
	Semideciduous	Primary	3,683,920.0	13,509,303	54,037,212	439,052,351
	tropical forest	Second growth	1,240,476.4	4,548,951	18,195,804	147,840,910
	Deciduous tropical	Primary	3,681,826.0	13,501,624	54,006,497	438,802,784
sma	forest	Second growth	2,898,742.8	10,629,980	42,519,919	345,474,344
ste	Woody xerophytic	Primary	921,651.4	3,379,788	13,519,151	109,843,103
osy	shrubland	Second growth	68,730.1	252,040	1,008,161	8,191,305
Ē	Non-woody	Primary	259,937.9	953,218	3,812,873	30,979,590
	xerophytic shrubland	Second growth	- 100,410.0	- 368,214	- 1,472,854	- 11,966,939
	Other woody	Primary	10,950.9	40,158	160,632	1,305,131
	vegetation types	Second growth	- 4,678.6	- 17,157	- 68,628	- 557,600
	Other non-woody	Primary	- 39,498.7	- 144,846	- 579,383	- 4,707,487
	vegetation types	Second growth	ND	ND	ND	ND
	Woody hydrophytic	Primary	755,801.4	2,771,599	11,086,397	90,076,974
	vegetation	Second growth	100,605.0	368,929	1,475,715	11,990,184
	Non-woody	Primary	25,805.1	94,630	378,519	3,075,467
	hydrophytic	Second growth	ND	ND	ND	ND
	Grassland	NA	8,439,288.8	30,947,716	123,790,864	1,005,800,768
	Forest plantation	NA	72,251.4	264,953	1,059,812	8,610,971
	Country-wide total		48,258,880	176,970,137	707,880,548	5,751,529,454

 Table 3. Potential monetary value of the Carbon sequestration services provided by Mexican ecosystems when traded in a voluntary forest C market or when valued in terms of the Social Cost of Carbon. See explanation in the text.

Uncertainties and possibilities for future improvement

This is the first ecosystem service account developed in Mexico according to the SEEA-EEA framework. Given the potential relevance and applications of its results (see below), it is important to entertain how accurate and reliable they are.



First, the georeferenced field data from the National Forest Inventory were the basis for estimating C density and between-dates differences in C stocks. The sampling methods used for collecting the data in the field and the procedures used for processing the data are fully documented (CONAFOR, 2012), have been extensively tested, and their implementation in the field is routinely controlled for quality (SEMARNAT, 2004). The NFI dataset is the official national database that provides information on the size, spatial distribution and condition of forest resources (SEMARNAT, 2004). This information is used to support the development and implementation of national policies related to forestry activities; it is also the basis for Mexico's report to FAO's Forest Resource Assessment (*e.g.*, FAO, 2010) and for Mexico's National GHG Emissions Inventory submitted to the UNFCCC (SEMARNAT-INECC, 2018), and has been often used as input and reference data for modelling forest C stocks through remote sensing methods (*e.g.*, Saatchi *et al.*, 2011; Cartus *et al.*, 2014; Rodríguez-Veiga *et al.*, 2016). Attempts to estimate the magnitude and spatial distribution of forest biomass stocks in Mexico through remote sensing data and statistical models have not been yet entirely successful as they show significant discrepancies both, with the NFI field data and with each other (Rodríguez-Veiga *et al.*, 2016); their use was, therefore, not entertained.

The NFI field data were then combined with INEGI's Vegetation and Land Use charts and the CEC's map of North American Terrestrial Ecoregions (level II) to obtain regionalized estimates of C density (*per* ecosystem type per ecoregion) and to map these estimates over the whole country. INEGI's charts are constructed through a combination of visual interpretation of optical imagery and field verification to create a country-wide land-use and vegetation class vector layer at a scale of 1:250,000, with a minimum mappable area of 25ha (INEGI, 2009). This imposes some limitations on the accuracy of the results, as vegetation/land-use patches smaller than 25ha embedded in a matrix of a different vegetation type or land-use class are overlooked, thus ignoring a potentially important source of heterogeneity when spatially extrapolating the C density estimates. Similarly, although the horizontal scale of the North American Terrestrial Ecoregions Level II map is not formally stated, it is described for use at national level and to be displayed at coarse resolution (approx. 1:10,000,000). The coarse horizontal resolution of the two maps used for spatializing the C density estimates imposes additional limitations on the use of the results obtained. Such limitations might not be noticeable/significant when looking at total figures for the whole country but will certainly be significant when trying to disaggregate the results at lower levels, such as State- or municipal-level.

Our estimates are completely consistent with the C sequestration rates and C stocks reported in the National GHG emissions Inventory for the LULUCF sector that is submitted by the Mexican government to the UNFCCC in its national communications. This is an advantage of using —as stated in the **General considerations on data and methods** (see above) — only official data or data used by official entities, as well as transparent replicable modelling methods for these estimations. However, in this study we were unable to report on the stocks and emissions of C from soil organic carbon due to the lack of suitable data, as discussed in Annex A. On-going efforts by INEGI to use advanced statistical models as well as biophysical models (*e.g.* the S-World approach) to exploit the vast set of soil data that they have compiled in the field over the last 50 yr to map soil organic C for different points in time might help to fill this important data gap.

Thus, although this analysis was based on the best datasets available for the country, these are not perfect and impose some limitations on the further use and interpretation of the resulting accounts, particularly with regard to the extent to which the results, maps and figures obtained can be meaningfully used at the local scale (State- or municipal-level). It would be desirable to carry out an independent uncertainty analysis of the results but no other, more accurate data set is currently available; a possibility on this regard would be using the field data from the third sampling cycle of the National Forest Inventory, which is to be completed in 2020, to do so. The next iteration of this study



could be improved significantly particularly if a higher resolution vegetation and land-use map becomes available. INEGI is currently working on improving its methods and procedures and the next iteration (series 7) of its vegetation and land-use chart, due to be published in 2021, might help to improve the accuracy of both the C estimates and the C maps. The need to stratify the NFI field data by ecoregion based on the CEC map would be unnecessary were the NFI's sample to be significantly augmented so that local estimates of biomass and C density could be obtained. However, due to the enormous additional investment that this would require, this does not seem feasible in the near future.

Potential policy relevance

Results from this initial pilot study can support policy implementation and decision-making in Mexico in several ways:

First, the pilot study and the results obtained are useful by themselves as the General Law on Sustainable Forest Development mandates SEMARNAT to "... define the methodologies for the valuation of environmental goods and services provided by forest ecosystems..." and CONAFOR to "... propose the valuation of environmental goods and services of forest ecosystems, according to the methodologies defined by [SEMARNAT]...". The Law also states that "... the valuation of environmental goods and services..." is a mandatory criterion of the country's forest policy and that CONAFOR must compile "... the National Forest and Soil Inventory, which must systematically integrate the statistical and accounting data of environmental goods and services...". Thus, the possibility of proposing that the methods developed for this pilot exercise be adopted officially by SEMARNAT and the results from the forest C account be integrated by CONAFOR as part of the National Forest and Soil Inventory could be explored.

On the other hand, the results from this initial pilot study could inform and refine the implementation of environmental management policies, instruments and government programs such as the Payment for Environmental Services and the watershed restoration programmes run by CONAFOR. For example —as recognized by CONAFOR officials— they could provide additional information to refine the targeting of and evaluate the costs of opportunity in such payments, as well as to evaluate the cost/benefit of restoration projects.

Maintaining carbon stocks in Mexican forests by reducing carbon losses from degradation and deforestation is a critical component of climate change mitigation. Results from this study show, in detail, that the country's forest ecosystems function as a net sink of C and highlight the important role that they play in fulfilling the mitigation goals pledged by the Mexican government as part of its Nationally Determined Contribution in the Paris Agreement. In addition, a sound analysis of how carbon stocks vary in relation to environmental conditions and in relation to land use and human activities is necessary for these purposes. Unlike the current LULUCF emissions inventory compiled by CONAFOR, our analysis is spatially explicit, which provides important additional information (although with some limitations as discussed in the previous section) as the resulting maps clearly depict and help to identify the zones where C sequestration and C stocks are the highest, as well as the ecosystem types that are most valuable in this regard. This could then facilitate targeting or prioritizing those particular zones or ecosystems in conservation programmes, incentives for improved land-use practices, etc.

If this information were made available to the general public, it might encourage conservation actions by local actors (*e.g.*, companies, NGOs, civil society groups, municipal governments, etc.). Hence, this would help climate change and forest conservation policies that are traditionally designed and implemented by the federal government, to gradually transition to local-level actions.



Developing a consistent time series is essential to fully show the potential of the carbon account. If replicated consistently over time, the C account could be used to monitor progress towards Mexico's climate change mitigation obligations as well as the progress and effectiveness of conservation, forest management, and other measures.

8.2 Water supply

Water used for human consumption, agricultural irrigation, electricity generation and other purposes that contribute to the welfare of society, can be pumped out from underlying aquifers (groundwater) or drawn from water bodies such as lakes, rivers and streams (surface water). Although ecosystems do not properly create water, they do modify the amount and properties of water moving/passing through them and, therefore, the provision of fresh water for anthropogenic uses can be rightly considered as an ecosystem service.

Although all components of ecosystems, from microbes to top predators, can affect the water provision service, vegetation is usually the most important one. Overall, the major effect of ecosystems is to reduce the volume of water available because vegetation consumes water through transpiration. If water is not limiting, taller vegetation will consume water at higher rates than shorter vegetation. Vegetation with deeper roots has greater access to soil moisture. Thus, in water-scarce environments, deeply rooted plants can successfully tap into a scarce water source. Arboreal vegetation will usually utilize more water than shorter vegetation because of its height and rooting depth. Tall, structurally complex vegetation shows increased gas exchange and thus transfers water to the atmosphere more rapidly than short, smooth vegetation. As a consequence, land-use and land-cover changes affect patterns of evapotranspiration and water infiltration and retention, thus altering the entire hydrologic cycle and affecting the ecosystem service of water supply as well.

Modelling the interrelationships between ecosystems, landscape changes and water supply is not easy. Complex hydrological models of such connections and associated processes do exist (*e.g.*, the <u>Soil</u> and Water Assessment Tool, the <u>Hydrological Simulation Program-FORTRAN</u>, the <u>Water Evaluation and</u> <u>Planning</u> system, etc.) but all of them are data- and time-intensive and therefore difficult to apply in datapoor countries/areas. There is, thus, a need for simpler models that can be more easily applied, especially when examining several ecosystem services, where comparisons in relative terms may suffice. For this reason, for this pilot study I explored two indirect, but spatially explicit, methods for which data are readily available, as described below. The pilot study was carried out with data and technical support from both the Dirección de Recursos Naturales, INEGI and Dirección General de Estadística e Información Ambiental, SEMARNAT.

Water yield

Water yield is simply defined as the fraction of precipitation that is not returned to the atmosphere through either evaporation or transpiration and, therefore, represents the supply of water for all potential uses. For a given landscape, water yield is given by the relationship:

$$Y = \left(1 - \frac{AET}{Pp}\right) \cdot Pp$$

Where Y is water yield, that is, the annual amount of rainfall that is not lost back to the atmosphere through evaporation and transpiration; AET is actual evapotranspiration and Pp is annual precipitation. This formulation is unable to capture the exchange between surface water and groundwater via infiltration; it lumps together groundwater and surface-runoff. The relationship AET/Pp —that is, the fraction of precipitation that undergoes evapotranspiration— is determined by climate, the



soil water storage properties and vegetation cover, as described by the Budyko curve (see <u>Zhang *et al.*</u>. <u>2004</u>).

In order to model water yield in a spatially explicit manner, it can be assumed that water yield can be reasonably approximated solely by the local interaction between precipitation and evapotranspiration (Mendoza *et al.*, 2011) and, thus, the equation above can be applied on a *per* pixel basis to estimate the amount of water running off each pixel; this value represents the pixel's contribution to the annual average water yield of its drainage basin. Mean annual precipitation data were obtained from the reference climate dataset (average over the period 1961-2000; 30" spatial resolution) compiled by the Mexican Meteorological Service (available in http://atlasclimatico.unam.mx/AECC_descargas/). Due to the paucity of AET data for Mexico, I used instead the MODIS Global Terrestrial Evapotranspiration product (NASA MOD16A2) developed by the University of Montana and regularly released by NASA. This product provides 500m resolution, 8-day estimates of actual evapotranspiration, potential evapotranspiration, latent heat flux and potential latent heat flux from the world's vegetated land areas. These variables are estimated with an *ad hoc* algorithm based on the Penman-Monteith equation (Monteith, 1965), using other MODIS data products (viz., Land cover, Leaf Area Index, Albedo, Photosynthetically Active Radiation) and meteorological data (air pressure, air temperature, humidity, radiation) (see Mu et al., 2013). MOD16A2 data are available for the period starting on the first week of 2000 until today. For this study, I used the datasets corresponding to 2002, 2007, 2011 and 2014, and summed them up as appropriate to obtain estimates of annual actual evapotranspiration; annual precipitation was kept constant at the 1961-2000 average values. The data sources and procedures used for these purposes are fully described as Procedimiento 4.2 in Annex B.

Fig. 9 shows the geographic variations in (estimated) water yield (in m³) across the country for the four selected years. Each pixel's value in these maps represents the contribution of that parcel to the annual water yield of its own drainage basin and, thus, it has no meaning beyond the drainage basin where it is located (see below). Nevertheless, these maps portray the heterogeneity and geographic distribution of water yield as affected by local factors and conditions such as precipitation, temperature regime, vegetation type, etc. Fig.9 shows that the Southern part of the coastal plain of the Gulf of Mexico, together with the Lacandon mountain range and other mountainous areas in the State of Chiapas are the parts of the country where the highest amount of water is available. By contrast, the leeward slopes of the Eastern Sierra Madre and the eastern half of the Mexican Plateau are the regions where less water is available. The maps in Fig.9 also reveal one important limitation of this method: the MOD16A2 product utilizes a MODIS-based estimate of Leaf Area Index (LAI) as one of its inputs. For this reason, AET cannot be estimated for water bodies, zones devoid of vegetation or scarcely vegetated areas, for which LAI is zero or undetectable by the MODIS sensors (notice the blank spots in the maps in Fig.9), thus leading to an underestimation of water yield.





Fig.9. Water yield per pixel (m³) in Mexico as of 2002 (top) and 2007 (bottom). See explanation in the text.




Fig.9 (cont.). Water yield per pixel (m³) in Mexico as of 2011 (top) and 2014 (bottom). See explanation in the text.



The spatial representation of the estimated water yield values allows readily relating those to the spatial distribution of ecosystems and land-use/land-cover patterns. Overlaying these maps with the maps of Mexico's ecosystems (for the corresponding dates; see Fig.2d above) allows examining how land-cover/land-use affect water supply across the landscape, the relationship between the country's ecosystems and water supply, and also evaluating to what extent the different ecosystem types contribute to the amount of water potentially available for various uses. Fig. 10 below shows the country-wide estimates of annual water yield per ecosystem type for 2002, 2007, 2011 and 2014. Water yield in each ecosystem type in a given year depends on the surface area it covers, the precipitation and evapotranspiration in that area, and the effect of land cover on runoff and infiltration. Grasslands (including natural grasslands plus pasturelands) are the ecosystems that most contribute to water yield, followed by evergreen tropical forests and coniferous forests (particularly those along the Western Sierra Madre). The much smaller contribution of arid and semiarid ecosystems (*i.e.*, xerophytic shrublands) to water yield is also evident.

Fig. 10 also shows how the contribution of the different ecosystems to water yield varies over time. However, these changes are not easy to interpret as these estimates respond to, on the one hand, changes in the extent and/or condition of ecosystems and in land-use patterns. On the other hand, however, the estimates are also influenced by between-years fluctuations in precipitation and temperature, as these affect AET —as recorded by the MODIS data. The relatively large between-years variations observed in Fig. 10 suggest that the effect of such weather fluctuations might be larger than — and obscure— those due to changes in the ecosystems' extent/condition. In fact, for this same reason I found a number of data points for which the estimated AET in a given year was higher than the 1961-2000 average annual rainfall (thus leading to negative values for the water yield estimates); in those cases, I set the actual evapotranspiration equal to precipitation and, thus, water yield equal to zero.



Fig. 10. Country-wide annual water yield per natural ecosystem type for 2002, 2007, 2011 and 2014. Notice the different scale of the vertical axis in the top and bottom charts. See explanation in the text.



One other important shortcoming of this approach is that water yield encompasses all water (ground- and surface water) potentially available for use. This makes it difficult to relate water supply with water demand, as the demand/abstraction data are disaggregated per water source and by either drainage basin (for surface water) or aquifer (for ground water).

Surface water supply

The second method tested aims to estimate the supply of surface water only. It is an improved adaptation of the methods described in the Official Mexican Standard NOM-011-CNA-2015 *Conservación del recurso agua - Que establece las especificaciones y el método para determinar la disponibilidad media anual de las aguas nacionales* issued by the National Water Commission (CONAGUA). The method starts with a simple conceptual model of water balance in a hydrological basin:

Pp = AET + ES + I

In words, out of the total amount of rain (Pp) that falls in the basin, a fraction returns to the atmosphere by direct evaporation from the soil's or plants' surfaces as evapotranspiration (AET), another fraction (I) infiltrates the soil and can potentially reach the underlying aquifer; finally, another fraction (ES) runs off over the soil's surface and can reach water bodies and water courses.

The ES fraction is the natural runoff in the basin, that is, the "...mean volume of surface water that is annually collected by the basin's natural drainage network... the average amount of freshwater that runs off in an unregulated watershed..." (NOM-011-CONAGUA-2015). ES can be directly computed from hydrometric records from gauged streams. However, in the absence of sufficient field data, the NOM-011-CONAGUA recommends using the precipitation-runoff method to indirectly estimate the average volume of annual natural runoff (ES). This method states that:

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Average annual natural runoff volume in the basin =
Average annual Pp * Runoff coefficient * Basin area
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The runoff coefficient (CE) is the fraction of rain that falls in the basin and does not return to the atmosphere through evapotranspiration, nor does it infiltrate the subsoil. The runoff coefficient is determined by the nature of the substrate and the vegetation cover in the basin, which together determine the possibilities of water infiltration to the subsoil and/or that the rain becomes intercepted by vegetation and then lost by direct evaporation or absorbed by plants and eventually transpired to the atmosphere.

The NOM-011-2015's indirect method provides lookup tables for estimating CE values based on (roughly defined) relative levels of substrate permeability and soil cover interception capacity; based on these, basin-level, tabular estimates of ES are computed. In order to obtain more refined, spatially explicit estimates of ES, for the pilot tests of the NCAVES-Mexico project we adapted the indirect precipitation-runoff method as follows: First, we used the ranks of relative substrate permeability and soil cover interception determined by INEGI experts. INEGI's scheme is based on a much more detailed analysis of the permeability properties of both, the underlying geological substrate and the soil, as well as of the possibility of rainfall interception by vegetation depending on its density and structure. In addition, INEGI's ranks are determined in a spatially explicit manner using INEGI's 1:250,000 soils, geology and vegetation and land-use charts. Secondly, based also on the assumption that surface runoff can be reasonably approximated solely by the local interaction between precipitation, evapotranspiration and infiltration (Mendoza *et al.*, 2011), we applied the equation above at the pixel level (rather than for the entire hydrological basin) to estimate the contribution that each pixel makes to the basin's yearly natural runoff:



A pixel's contribution to the average annual natural runoff in the basin = Average annual Pp * Runoff coefficient * Pixel area

The data sources and procedures used for this purpose are fully described as Procedimiento 4.3 in Annex B.

Fig. 11 below shows the geographic variations in natural surface runoff (*i.e.*, the amount of rain water contributed by each pixel to the natural runoff in the basin) in Mexican ecosystems for 2002, 2007, 2011 and 2014. As can be seen, the Southern part of the coastal plain of the Gulf of Mexico, together with the Lacandon mountain range and other mountainous areas in the State of Chiapas are the parts of the country where the highest amount of surface water is available. By contrast, the leeward slopes of the Eastern Sierra Madre, all of the Mexican Plateau and most of the Baja California peninsula are the regions with the smallest supply of surface water. As the model was built using the recent long-term (1960-2000) average precipitation values, differences between dates in surface runoff are only due to changes in the extent and/or condition of ecosystems and these are not very evident in the maps at the resolution used for these figures.

By overlaying these maps (Fig.11) with the corresponding maps of Mexico's ecosystems (see Fig.2d above), surface runoff can be readily related to the spatial distribution of ecosystems and land-use/land-cover patterns and thus examine to what extent the different ecosystem types contribute to surface water supply. Fig. 12 below shows the country-wide estimates of natural surface runoff per ecosystem type for 2002, 2007, 2011 and 2014. In this model, surface water supply by each ecosystem type in a given year depends only on the surface area it covers and its condition, as these affect runoff and infiltration. Grasslands (including natural grasslands plus pasturelands) are the ecosystems that most contribute to surface water supply, followed by evergreen tropical forests and coniferous forests. The much smaller contribution of arid and semiarid ecosystems (*i.e.*, xerophytic shrublands) to surface water supply is also evident. Fig. 12 also shows how the contribution of the different ecosystems to surface water supply varies over time. As the model was built using the recent long-term (1960-2000) average precipitation values, between-years differences are only due to changes in the extent and/or condition of ecosystems and in land-use patterns.

The model's outputs shown in Figs.11 and 12 can also be related to the corresponding outputs from the water yield model (Figs. 9 and 10). The water yield values are consistently higher (or much higher) than the corresponding values of natural surface runoff as the former encompass both, surface and ground water. Those parts of the country where both models (cf. Figs. 9 and 11) yield not-sodifferent values (e.g., the Southern part of the coastal plain of the Gulf of Mexico) are areas where most of the water yield comes as surface runoff and a relatively small fraction infiltrates the soil to potentially become groundwater. Vice versa, those parts of the country where water yield is much higher than surface runoff (e.g., the western half of the Mexican plateau and many areas in the Yucatán peninsula) should be areas where a large fraction of the rain that is not lost through evapotranspiration infiltrates the soil to become groundwater. In arid and semiarid ecosystems (*i.e.*, xerophytic shrublands), water yield (Fig.10) is much higher than surface runoff (Fig.12), thus indicating that a larger fraction of water infiltrates the soil; by contrast, in ecosystems dominated by hydrophytic vegetation, a larger fraction of water yield comes as surface runoff. While it is tempting to calculate the difference between the corresponding maps in Figs. 9 and 11 to obtain a pixel-level estimate of groundwater supply (as well as the difference between Figs. 10 and 12 to estimate the volume of groundwater contributed by each ecosystem type), the shortcomings of the two models and the lack of an independent data set to validate the estimates that would be thus obtained do not warrant the soundness of such exercise.



Escurrimiento por pixel 2002 (m3)



Fig.11. Natural surface runoff per pixel (m³) in Mexico as of 2002 (top) and 2007 (bottom). See explanation in the text.





Fig.11 (cont.). Natural surface runoff per pixel (m³) in Mexico as of 2011 (top) and 2014 (bottom). See explanation in the text.





Fig. 12. Country-wide annual natural runoff per natural ecosystem type for 2002, 2007, 2011 and 2014. Notice the different scale of the vertical axis in the top and bottom charts. See explanation in the text.

While useful for examining the relationship between water supply and land-use/land-cover patterns, the pixels' values portrayed in the maps in Fig.11 actually represent the contribution that each parcel makes, annually, to the natural surface runoff of its own drainage basin; an individual pixel's value has no meaning beyond the drainage basin where it is located. More importantly, these pixel-level maps cannot account for the between-pixels flows that produce surface and subsurface water flows, drainage and discharges to rivers, streams, creeks, etc. and, thus, should not be used for analysing hydrological processes. Therefore, in a second step, we added up the individual pixels' contributions as appropriate to obtain estimates of annual natural surface runoff for each of the 757 drainage basins that have been delineated in the country (Fig.13a). Such basin-level analysis can then be used to compare —and, to some extent, validate— the model's output with the natural surface runoff values independently obtained by the National Water Commission (CONAGUA, for its acronym in Spanish) either from gauged streamflow data or otherwise estimated (Fig.13b).

As can be seen, the indirect method is able to recover the overall pattern of surface runoff across the country, correctly identifying those basins where water runoff is the highest (*i.e.*, the Southern half of the Eastern Sierra Madre, the Southern Sierra Madre and the Western Sierra Madre), as well as those others where it is the lowest (*i.e.*, most of the Baja California peninsula, the leeward slopes of the Eastern Sierra Madre, and most of the Mexican plateau). However, the indirect method underestimates surface water supply in most basins, with one major exception: The Yucatan peninsula, for which the indirect method estimates a volume several orders of magnitude higher than the values reported by CONAGUA. The whole of the Yucatan Peninsula is a flat-lying karst landscape, composed of carbonates and soluble rocks (mostly limestone) and, due to its extreme karst nature, most of the rain falling there rapidly infiltrates the substrate to reach the underlying aquifer. In fact, no rivers occur in the northern half of the



peninsula (<u>Encyclopaedia Britannica</u>). The indirect method seems unable to capture the extremely high infiltration occurring in this part of the country, thus leading to a gross overestimation of surface runoff.



Fig.13. Natural surface runoff per drainage basin (m³), as estimated by the indirect method for 2014 (13a, top panel) compared to CONAGUA's estimates for 2016 (13b, bottom panel). See explanation in the text.



Water supply and water provision

The spatially-explicit model we developed based on the NOM-011-CONAGUA's precipitationrunoff method estimates the volume of surface runoff from each parcel (pixel) on the landscape; this represents the supply of water for all potential uses. However, under the SEEA-EEA framework, the benefit (in terms of human well-being) from this ecosystem service is only realized when water is actually used. Therefore, what should be valued is only the fraction of the total water supply that is actually used —that is, the provision of or demand for surface water.

Thus, the demand/provision/use of surface water supply should be determined. This step is usually based on socioeconomic, management and other similar data documenting the level of supply that is actually demanded or used by people in the study area. For this pilot study, we used the data on annual volume of surface water abstraction per drainage basin (Fig. 14a) that CONAGUA compiles as part of its process to determine surface water availability in the country's hydrological basins (DOF, 2016). The water abstraction data are, in turn, derived from the water concessions issued by CONAGUA, as recorded by the National Public Registry of Water Users (REPDA, for its acronym in Spanish). The concessions include water for consumptive (*i.e.*, agricultural, public service, industrial and electricity generation by thermal power plants) and non-consumptive (*i.e.*, electricity generation by hydroelectric power plants) uses (note that the consumptive use of water by rain-fed crops is implicitly accounted for by the runoff coefficient, CE, of the model). The water concessions data represent the maximum volume of water that the user can abstract annually, rather than the volume of water that is actually abstracted every year.

Fig. 14a shows the volume of surface water concessions issued by CONAGUA for the 757 hydrologic basins delineated for the country. These data can then be compared with the basin-level surface water supply values estimated by the model (Fig. 13a) to produce a (partially-) spatially delineated balance of surface water supply/provision (see Fig.14b). This approach gives an indication of the over- or underutilization of this ecosystem service. As can be seen in Fig.14b, in most of the basins, the estimated volume of surface water supply is higher than the maximum allowable rate of water abstraction. The few basins in which the rate of water abstraction is much higher than their water supply are associated with big dams and/or electricity generating power plants: The Chicoasén dam in the State of Chiapas and the Miguel Alemán dam in the State of Oaxaca are hydroelectric power plants that make a non-consumptive use of enormous volumes of water; the Bajo Balsas basin, in the State of Guerrero, encompasses the Petacalco thermal power plant which makes a consumptive use of enormous volumes of water for cooling.

As shown in the previous section, the spatially explicit, pixel-level model developed for this study (Fig.11) allows readily examining the relationship between ecosystems and land-use patterns and the supply of surface water. However, the spatial relationship between water supply and water use/demand is—in most cases—much more complicated. For example, water users are often located in places (usually within the same basin) different than the ecosystems that contribute to supplying the water; users often utilize water supplied by various ecosystems located upstream within the same basin; water supplied is used for various purposes (*e.g.*, domestic, irrigation, services or industrial uses) in different parts of the basin, etc. This lack of spatial concordance between the location of the users of this ecosystem service and the location of the ecosystems that supply the service means that water provision —as opposed to water supply— should only be reliably examined at the landscape or basin-level and it would prove very difficult, in practice, to reliably quantify and value the contribution of each ecosystem asset (and of each ecosystem type) to the ecosystem service of water provision.





Fig.14. a) Total volume (m^3/yr) of surface water concessions issued by CONAGUA per hydrological basin, as of 2016 (top); b) Difference between the basin-level surface water supply (m^3/yr) estimated by the model and the volume of surface water concessions (bottom). See explanation in the text.



Potential approaches for the monetary valuation of water provision services in Mexico

The resource rent and the replacement costs approaches are two methods potentially appropriate for the economic valuation of water provision services (UN, 2014). The replacement cost approach is based on the cost of replacing the ecosystem services from alternative sources (*e.g.*, desalination, use of recycled water, purchasing water from other areas, etc.). To estimate the replacement cost, goods that are traded in real markets and that can operate as substitutes for the non-market service to be valued should be identified and their necessary quantities and prices should be calculated or estimated. The method assumes that (i) if the service was lost it would be necessarily replaced by users, and that (ii) users will not change their pattern of use in response to a price increase. This approach has been recently utilized by López-Morales & Mesa-Jurado (2017) for the economic valuation of groundwater provision in central Mexico.

On the other hand, water supplied into Mexico's economy, as valued by CONAGUA, is the end result of a combination of fixed capital (reservoirs, water mains, piping, pumps, etc.), labour and other inputs, in addition to the ecosystem service of water supply itself. Hence, the value of the water provisioning service is not the same as the value of the water supply. The resource rent approach aims to deduct the costs of such inputs, to reach a residual that reflects the ecosystem's contribution. However, in some cases, the price paid by the service only covers the costs of supplying it to customers and, thus, the residual thus calculated turns out to be very small or even negative. An additional complicating factor is the fact that the price of water in Mexico is not only regulated by the government but is also subject to major distortions, *e.g.*, water for irrigation purposes is charged a 0 fee; electricity for agricultural irrigation is also subsidized, etc. Because of this, the resource rent method might not be feasible for Mexico's conditions, as it could lead —as has been the case in other countries (*e.g.*, the Netherlands, see Edens & Graveland, 2014)— to negative rents.

Uncertainties and possibilities for future improvement

A key feature of the SEEA-EEA framework is the requirement for ecosystem services to be first quantified in physical terms, as the economic valuation of their supply and use necessarily depends on reasonably sound information in physical terms. Thus, before applying economic valuation methods, the outputs from ecosystem service models should be validated. Otherwise, it is unknown how well the model describes the study area and how accurately evaluates the supply of services. Even if the outputs from a given model are not accurate enough, they can still be used but only in relative (increase or decrease; high or low values, etc.) rather than absolute terms.

The empirical model developed for this study does take into account the spatial distribution of precipitation, ecosystems and land-use types in Mexico. Because of this, a comparison of the model's outputs with the corresponding values independently computed/estimated by CONAGUA showed (Fig. 13) that the model successfully recovers the overall pattern of surface runoff across the country both at the pixel- and the drainage basin level. However, the model's predictions underestimate surface water supply in most basins and, in some cases, fails to adequately capture the interactions between infiltration and surface runoff. The model seems unable to reasonably capture the high infiltration that occurs in the extreme karst landscape of the Yucatan peninsula, thus leading to a serious overestimation of surface water supply therein.

In conclusion, this rather simplistic modelling approach does provide useful information about the spatial distribution of surface runoff; about how the ecosystems, land-use and land-cover patterns in the country are related to the supply of surface water; and on how this changes over time in response to changes in the extent and condition of ecosystems and land use. However, the model's outputs can, at best, be used only in relative terms but their absolute values should be interpreted with caution. This



result suggests that more detailed, process-based models might be necessary to properly describe, in a spatially-explicit manner, the relationship between ecosystems and water supply in Mexico. It would also be important to examine both components of water yield (*viz.*, groundwater and surface water), as they are inextricably linked, and play major —but varying— roles in the water supply service in different parts of the country. Complex hydrological models that better capture the interrelationships between ecosystems, landscape changes and water supply and associated processes do exist (*e.g.*, the <u>Soil and</u> <u>Water Assessment Tool</u>, the <u>Hydrological Simulation Program-FORTRAN</u>, the <u>Water Evaluation and</u> <u>Planning</u> system, etc.). Unfortunately, the much more detailed data that they demand are not yet available for Mexico.

The model developed provides spatially-explicit estimates of the supply of surface water for all potential uses but, under the SEEA-EEA framework, the benefit from this ecosystem service comes only from the fraction of the total water supply that is actually used —that is, the provision of or demand for surface water. Georeferenced administrative records maintained by CONAGUA were used to try to estimate the demand for surface water. However, the insufficient detail of the water users' data (e.g., the user's reported location does not necessarily correspond to the physical location where water is abstracted/used) and, above all, the spatial disparity between the location of water users and the location of the ecosystem(s) supplying water made it very difficult to discern the relationship between ecosystems and land-use patterns and the provision of surface water. As a result, our analysis of water supply – water demand was greatly oversimplified. For each drainage basin, only a single value denoting the total volume of annual abstraction allowed in surface water concessions— was used. Those basin-level values are clearly insufficient to represent the complex, multiple aspects of water use and allocation among uses within the basin. For example, the use of a basin-level value of water use implicitly assumes that water consumed is drawn equally from every pixel within the basin but, in reality, some point source intakes (*e.g.*, those serving urban areas) may account for a much larger portion of the water demand while others may abstract much smaller volumes; water demand may differ greatly between parcels of the same ecosystem type; water transfers (*e.g.*, for irrigation), either between sub-basins or between years (through dams), can obviously not be captured by the model.

An alternate modelling approach based on information derived from remote sensing data was also tested. Although this model's outputs are promising, they are affected by important issues such as the limited capability to detect and estimate LAI in sparsely vegetated areas (causing significant data gaps) and, more importantly, their responsiveness to short-term (interannual) fluctuations in temperature and rainfall which would obscure the effect of changes in the extent and condition of ecosystems and land-use patterns...which are the main focus of the study.

Potential policy relevance

The spatially-explicit model developed for this pilot study utilizes data that are readily available; takes into account the spatial distribution of precipitation, ecosystems and land-use types in Mexico; and successfully captures the overall pattern of surface runoff across the country both at the pixel- and the drainage basin level. Thus, despite its limitations (discussed in the previous section) this model can be used as an initial step (say, tier 1) in the design of suitable methods for the valuation of the water supply services provided by forest ecosystems in Mexico (part of SEMARNAT's mandate). As additional, better data become available, the same method can be further refined and its limitations overcome, or other more complex, data-demanding, process-based, hydrological models can be implemented.

Even though the model's outputs were not accurate enough, they can still be used in relative terms, providing additional, spatially-explicit information to refine the implementation of environmental management schemes run by CONAFOR such as the Payment for Hydrologic Environmental Services and



the watershed restoration programmes. The model's outputs could help refine the targeting of the payments and evaluate the cost/benefit of different restoration projects.

Albeit with some limitations, the model's outputs show how surface water supply varies in relation to environmental conditions, and with the extent and condition of ecosystems and land-use patterns. They also show where water utilized for anthropogenic uses comes from and how different ecosystem types contribute to water supply. This information would be particularly useful for identifying and locating both the specific ecosystems that supply the service (water for anthropogenic uses, in this case) in a given region, and its beneficiaries. This would allow making the beneficiaries aware of their dependency on those ecosystems and the importance of preserving them, perhaps setting the foundations for the design of schemes by which the beneficiaries provide material incentives to the ecosystem owners to protect the ecosystems that supply the water they benefit from.

More generally, the model's outputs could be useful for, for example, prioritizing key watersupplying zones or ecosystems in conservation programmes, incentives for improved land-use practices, etc.; for identifying potential unintended impacts of development or infrastructure projects and potential trade-offs between alternative land management decisions (*e.g.*, converting the land to agriculture, urban development or other land uses *versus* preserving the ecosystems contributing to water supply); for informing the design of restoration or management projects aimed at improving or maintaining key water-supplying ecosystems, etc. Were this information made available to the general public, it might encourage conservation actions by local actors (*e.g.*, companies, NGOs, civil society groups, municipal governments, etc.), to gradually transition from conservation policies solely designed and implemented by the government, to local-level actions.

8.3 Food crop production

Agricultural production, especially in intensive agricultural areas, is determined, to a large extent, by a variety of human inputs such as specific plant breeds, fertilizers and pesticides, water management, labour, land and other capital assets. All or most of these inputs are already accounted for in national accounts. In addition to these inputs, however, agricultural production also depends on a range of ecosystem services including water capture, abstraction of soil water, soil formation, nutrient cycling and soil nutrient uptake, pollination, pest control, etc. (UN, 2014). Some of these services (*e.g.*, soil water and nutrient uptake) are generated on the very same land used for agricultural production through natural processes such as nutrient and water cycling, soil biodiversity, etc. Other services (*e.g.*, pollination, pest control, etc.) may be generated somewhere else.

The various ecosystem services that contribute to agricultural production are defined in the Common International Classification of Ecosystem Services (<u>CICES 2018</u>) as: "The ecological contribution to the growth of cultivated, land-based crops that can be harvested and used as raw material for the production of food, non-nutritional purposes, or as a source of biomass-based energy". For accounting purposes under the SEEA-EEA framework, the natural processes that enable and contribute to agricultural production should —ideally— be identified and quantified to properly determine the ecosystem contribution. In practice, however, disentangling the contributions that ecological processes and human inputs make to the final output of agricultural production is difficult. Agricultural production services are therefore seen as a form of complex "co-production" by people and nature that is difficult to disaggregate (<u>CICES 2018</u>). For this reason, most of the studies on ecosystem service valuation simply use the volume of crop before harvest as the physical measure of the final ecosystem service, explicitly or implicitly adopting the 'harvest approach' described in the SEEA-EEA framework (<u>UN, 2014</u>); that is, assuming that the various flows (*e.g.*, pollination, nutrients from the soil, and water, etc.) that constitute inputs into the growth of the mature crop are in fixed proportion to the quantities of harvested product.



Disaggregating the contributions from ecosystems and human inputs might then be more feasibly done at the monetary valuation stage.

For the pilot study on food crops provision, the six priority crops that the Ministry of Agriculture (SAGARPA, for its acronym in Spanish) targeted in its *Sectoral Program for Agricultural, Fisheries and Food Development 2013-2018* were chosen: maize, beans, wheat, rice, sorghum and soy. The top priority status of maize, beans, wheat and rice has been reaffirmed by the new federal government administration for the term 2018-2024. Out of the several varieties of these crops that are grown in Mexico, only those meant for human consumption were included in the analysis (*e.g.*, white maize grain, palay rice, wheat grain, etc.). Following the arguments above, I used crop production (kg/ha/yr) as the indicator of the ecosystem service and expressed all crops in terms of weight at harvest —but realised that the value of the crops, both on a per kg and on a per ha basis, varies considerably between crops.

The study was based on official annual agricultural production data reported by the Ministry of Agriculture's Information office (SIAP) in its *Statistical Yearbook of Agricultural Production*. The yearbook provides tabular data for the area sown (ha), area harvested (ha), area damaged by natural hazards (ha), production volume (tm), yield (tm/ha), average price (\$/tm) and production value (thousand pesos) for each crop, per municipality, for each harvest cycle of the year. Data for the years 2007, 2011 and 2014 were chosen for analysis. The SIAP's tabular data are insufficient for the NCAVES-Mexico project as the SEEA-EEA framework demands spatially explicit estimates of ecosystem services supply. In order to spatialize the food crop production data, these were combined with the Municipal Geostatistical Framework and the Vegetation and Land-use charts (for the years 2007, 2011 and 2014) produced by INEGI. Using these charts, I assigned the annual production volume (or value) of each crop in a given municipality to those polygons that, according to the Vegetation and Land-use chart, were devoted to agriculture in that municipality. The vegetation and land-use charts distinguish rain-fed from irrigated crop lands and annual, semi-permanent and permanent crops, which made allocation of the production data closer to reality. Production data were allocated to the polygons devoted to agriculture (in the corresponding Vegetation and Land-use chart) as a function of the polygons' proportional area in the municipality. The data sources and procedures used for this purpose are fully described as Procedimiento 4.3 in the Anexo metodológico.

Figs. 15a-f below show the geographic distribution of production volume (in tm/polygon) of the six crops examined, as of 2007, 2011 and 2014. As can be seen, the geographic distribution and extent of most of the crops examined (except for soybean) do not change substantially or consistently over time: The same crops are grown in the same regions year after year. Perhaps the most noticeable feature is the broad geographic distribution of the two crops native to Mexico, maize and beans, which are grown practically throughout the country under a wide range of environmental conditions. Their vast genetic diversity, their long history of domestication under a very diverse range of environments, and their role as major food staples in Mexico, have led to the development of numerous cultivars, each of them especially adapted to particular environmental conditions (Gepts, 2014; Katz, 2018).

The other two food staples, wheat and rice, have a narrower geographic distribution: Wheat across the Mexican plateau and the areas of high-input agriculture on the coastal plain of the states of Sonora and Sinaloa. Rice is only grown in the more humid zones in the Southern part of the country or associated with wetland areas. Sorghum and soy have a mixed role, being used not only for human consumption but also as raw materials for animal food and other industrial uses. Thus, the geographic distribution of these crops seems to respond not only to environmental conditions but also, and to a large extent, to market conditions.









Fig. 15b. Production volume (in tm/polygon) of beans as of 2007, 2011 and 2014





Fig. 15c. Production volume (in tm/polygon) of wheat as of 2007, 2011 and 2014 $\,$









Fig. 15e. Production volume (in tm/polygon) of sorghum as of 2007, 2011 and 2014 $\,$







Another noticeable feature in Fig.15 is the striking difference in the yield obtained (for most of the crops examined) in those parts of the country (for example, the southern part of the State of Sonora, the El Fuerte valley in the State of Sinaloa, and the agricultural area of Ciudad Insurgentes-Ciudad Constitución in Baja California Sur) where high-input agriculture is practiced and production is devoted to the national and even international markets, compared to the yields obtained in most other parts of the country where crops are usually rain-fed, and production is mainly devoted to local markets and self-consumption.

Although the extent of agricultural ecosystems in the country has increased slightly from 2007 to 2014 (see Fig. 3b above), the variations observed (see Fig. 15) in the production volume of these crops over the same period are not —in most cases—correlated with changes in the cultivated area (see Table 4). Therefore, the changes over time in production volume might be mostly related to environmental conditions (*e.g.*, weather, pests, etc.) affecting yield, or to market demand, price or other circumstances that affect the farmers' decisions. Part of these effects can be inferred (see Table 4 below) from the difference between area sown and are harvested which, in some cases (*e.g.*, maize in 2011), are substantial and are usually caused by environmental hazards (*e.g.*, droughts, pests, diseases, etc.).

		2007	2011	2014	
	Area sown (ha)	1,318,651	1,507,279	1,097,680	
Maize	Area harvested (ha)	1,299,832	1,060,326	1,063,704	
	Production (tm)	9,291,484	6,364,770	8,131,011	
	Area sown (ha)	6,331,672	5,807,380	5,837,647	
	Area harvested (ha)	5,592,916	4,621,157	5,527,652	
	Production (tm)	12,485,966	9,509,013	12,579,873	
	Area sown (ha)	190,400	209,701	238,170	
	Area harvested (ha)	185,156	155,660	234,066	
Boon	Production (tm)	314,752	221,807	349,064	
bean	Area sown (ha)	1,498,077	1,296,333	1,535,827	
	Area harvested (ha)	1,304,086	739,312	1,446,832	
	Production (tm)	679,201	345,972	924,893	
	Area sown (ha)	532,552	599,373	596,275	
	Area harvested (ha)	531,744	589,866	593,101	
Who at	Production (tm)	3,190,824	3,553,390	3,429,625	
wheat	Area sown (ha)	173,127	115,491	116,757	
	Area harvested (ha)	159,935	72,356	113,510	
	Production (tm)	324,568	74,121	240,189	
	Area sown (ha)	25,740	20,702	28,360	
	Area harvested (ha)	25,710	19,292	28,093	
Pico	Production (tm)	150,810	122,246	184,268	
Rice	Area sown (ha)	47,796	16,109	12,719	
	Area harvested (ha)	45,239	14,745	12,550	
	Production (tm)	143,888	51,215	47,891	
	Area sown (ha)	493,871	672,025	645,995	
	Area harvested (ha)	486,302	660,217	643,244	
Corchum	Production (tm)	2,877,018	3,967,565	4,035,047	
Sorghum	Area sown (ha)	1,375,103	1,300,034	1,432,502	
	Area harvested (ha)	1,288,673	1,068,011	1,370,665	
	Production (tm)	3,325,902	2,461,746	4,359,010	
	Area sown (ha)	8,654	16,943	24,121	
	Area harvested (ha)	8,332	16,222	22,519	
Soy	Production (tm)	10,962	28,488	47,737	
	Area sown (ha)	64,703	149,776	187,410	
	Area harvested (ha)	54,248	139,290	183,110	
	Production (tm)	77,409	176,746	339,630	

Table 4. Changes over time in the country-wide area sown, area harvested and production volume of the six crops examined. Blue-shaded cells: Figures for irrigated crops; green-shaded cells: Rain-fed crops.



Potential approaches for the monetary valuation of food crop provision services in Mexico

As discussed in the *Country Assessment on Natural Capital Accounting and Valuation of Ecosystem Services in Mexico*, various approaches have been used for estimating (or imputing) the value of the contribution that ecosystem services make to agricultural production. As the value of production of agricultural, forestry and fisheries outputs is usually included in national accounts, valuation approaches used in these cases should focus on determining the contribution of the ecosystem service(s) to the market price of the product rather than on valuing the ecosystem service directly (<u>UN, 2014</u>).

The value of production of agricultural outputs includes the costs of fixed and produced capital (*e.g.*, land, machinery), labour and other human inputs (*e.g.*, fertilizer, irrigation, etc.), in addition to ecosystem services. For this reason, the market price of agricultural products cannot be used directly as an estimate of the value of the ecosystem service. The unit resource rent approach aims to identify and deduct the costs of such human inputs from the market price of agricultural products to isolate a residual —the unit resource rent— that reflects the value of the ecosystem contribution. The unit resource rent approach is usually recommended for provision services such as agricultural, forestry and fisheries outputs (<u>UN, 2014</u>), as suitable data on the value of benefits and inputs costs are usually available from the national accounts —albeit not at a disaggregated or spatially explicit level.

Figs. 16a-f below show the geographic distribution of production value (at market prices) of the six crops examined, as of 2007, 2011 and 2014 (in thousand current Mexican pesos). Variations in production value across the country in a given year are due to both, local differences in production volume (*cf.* Fig.15) and local variations in market price. This information might provide the basis for the monetary valuation of food crop provision services in Mexico.

Although decomposing the multiple inputs contributing to agricultural production seems to be a straightforward operation, it poses significant methodological and even conceptual challenges. For example, depending on the product destination, the value of food crop provision services can be estimated in terms of their contributions to either, the market value of commercial agricultural production or the utility value of subsistence agricultural production (<u>Nelson *et al.*</u>, 2011</u>). Also, if farmers can use some of the resources that contribute to agricultural production at no or reduced cost, then the residual unit resource rent will be very small or even zero, implying that the value of the ecosystem service is also very small or zero. Evidently, any agricultural subsidy in place would have a most significant impact on the results of a resource rent valuation. Thus, although the unit resource rent approach is often recommended for the monetary valuation of crop provision services, due consideration should be given to first determine whether this approach is appropriate for the local conditions.





Fig. 16a. Production value (in thousand Mexican pesos/polygon) of maize as of 2007, 2011 and 2014





Fig. 16b. Production value (in thousand Mexican pesos/polygon) of beans as of 2007, 2011 and 2014





Fig. 16c. Production value (in thousand Mexican pesos/polygon) of wheat as of 2007, 2011 and 2014





Fig. 16d. Production value (in thousand Mexican pesos/polygon) of rice as of 2007, 2011 and 2014





Fig. 16e. Production value (in thousand Mexican pesos/polygon) of sorghum as of 2007, 2011 and 2014





Fig. 16f. Production value (in thousand Mexican pesos/polygon) of soybean as of 2007, 2011 and 2014



Uncertainties and possibilities for future improvement

The empirical approach developed for this study in order to portray —based on tabular official statistics— the spatial distribution of food crop production in Mexico, does take properly into account the geographic distribution of ecosystems and land-use types in Mexico. Because of this, and as recognized by experts from the Servicio de Información Agroalimentaria y Pesquera–SADER (SIAP), the model's outputs successfully recover the overall pattern of food crop production (as for maize, bean, wheat, rice, sorghum and soybean) across the country.

However, as SIAP's data are aggregated at the municipality level, every polygon of a given landuse type (*viz.*, irrigated annual cropland, rain-fed annual cropland) within a given municipality was assigned the same production (volume or value) value, in proportion to its extent in the municipality. Thus, the models are unable to provide independent estimates for croplands within or outside protected areas, or for areas with varying levels of anthropic modification. Therefore, food crop production cannot be related to ecosystem condition and its changes over time.

Therefore, the simple modelling approach developed for this pilot study does provide useful information about the spatial distribution of the production of priority food crops in Mexico and on how this changes over time. However, the model's outputs (*viz.*, production figures for a given cropland area or polygon) are, in reality, imputed values rather than actual measurements or estimates. While the aggregation of all the values within a given municipality necessarily agrees with SIAP's municipality-level data, the individual figures (*i.e.*, for a given polygon) that the model provides can be used only in relative terms but their absolute values should be interpreted with caution.

Specific efforts to map agricultural production and thus complement Mexico's production statistics would be most valuable for future analyses on crop provision services. In fact, SIAP experts have been conducting efforts to examine, country-wide and in a spatially explicit manner, the location of and changes in agricultural production. However, the unique characteristics and dynamics of Mexico's agricultural sector, together with the extent and physical complexity of the country, impose important technical difficulties to map agricultural production with the desired temporal, spatial and thematic resolution. SIAP is currently working on two complementary approaches on this regard: First, a project aimed to map and monitor the main areas where maize, beans, sorghum and wheat are grown in Mexico. The project focuses on those parts of the country that concentrate 80% of the area sown with these four crops. Maps for the years 2013, 2014 and 2015 have already been completed and are in the process of being validated prior to their publication. The second initiative aims to delineate the country's agricultural frontier (distinguishing irrigated from rain-fed croplands) and monitor how this changes over time, based on the analysis and interpretation of satellite imagery. Three analyses have been completed to date: Series I delineates the agricultural frontier as of 2011-2012; series II corresponds to the period 2014-2015; the recently completed series III delineates the agricultural frontier as of 2017-2018. However, SIAP has not yet attempted to link the crop production data with these cartographic products.

Potential policy relevance

The empirical approach developed for this pilot study utilizes data that are readily available, takes into account the spatial distribution of ecosystems and land-use types in Mexico, and successfully captures the overall pattern of food crop production (as for maize, bean, wheat, rice, sorghum and soybean) across the country. Thus, despite its above-mentioned limitations this modelling approach can be used as an initial step (tier 1) in the design of suitable methods for the valuation of food crop provision services in Mexico (part of SEMARNAT's mandate). As additional, better data become available —such as



the mapping efforts being currently developed by SIAP— the same approach can be further refined and its limitations overcome.

Albeit with some limitations, the model's outputs can still be used in relative terms, providing additional information that can be useful to link, in a spatially-explicit manner, agricultural production with the ecosystems providing essential services such as water supply. This information would be useful for identifying and locating both the specific ecosystems that supply the service (water for agricultural irrigation, in this case) in a given region, and its beneficiaries. This would allow making the beneficiaries aware of their dependency on those ecosystems and the importance of preserving them, perhaps setting the foundations for the design of schemes by which the beneficiaries provide material incentives to the ecosystem owners to protect the ecosystems that supply the water they benefit from.

Conversely, given the importance of the agricultural sector in regional economic development, estimating the potential value of agricultural outputs from as yet unconverted natural habitat may also be of interest for land-use planning purposes as it represents production value foregone and, thus, it would help analyse the cost-benefit of (or trade-off incurred when) converting natural habitat to agricultural land.

8.4 Multiple ecosystem services

In the previous sections, the spatial distribution of the three ecosystem services hereby evaluated were individually (*i.e.*, one ecosystem service at the time) examined: Carbon sequestration (Fig. 8), surface water supply (Fig. 11), and food crop provisioning (Fig. 15). Ecosystems, however, supply multiple services simultaneously and, more importantly, the use of one ecosystem service usually affects —positively or negatively— the ecosystem's capacity to supply others. Examining the various services that ecosystems supply is important in order to value each ecosystem asset in its just dimension, and to identify possible synergies or —most likely— trade-offs between the use of different services. To obtain an approximate picture of the multiple services that Mexican ecosystem supply, a combined index was constructed by first expressing the supply levels of each of the three ecosystem services hereby considered, in relative terms (as a proportion of the maximum value reached as of 2007)¹, and then adding up the resulting values into a single, synthetic value. The index can take values from 0 (in a pixel where no Carbon is sequestered, no surface water is collected, and no crop is harvested) up to a theoretical maximum of 3 (in a pixel where all the three services reach the maximum values observed as of 2007).

Fig. 17a below shows the geographic distribution of the combined index of ecosystem services supply. The lowest values denote those places where low volumes of only one (or very low volumes of a combination of) ecosystem service(s) are produced, whereas highest values denote those parts of the country where large volumes of one (or a combination of several) ecosystem service(s) are supplied. The latter could be considered as the country's "hotspots", that is, those ecosystems that supply large volumes of one or more ecosystem services; these are, mainly, the coastal plain of Sonora-Sinaloa, the mountain ranges of eastern Oaxaca, the Los Tuxtlas mountain range, and the mountain ranges (northern, southern and Lacandon) in the State of Chiapas.

One difficulty with the interpretation of the map in Fig.17a is that a given value can result from a certain level of one single service or from the combination of various levels of different services, and the

¹ Food crop provision actually comprises six crops (*viz.*, maize, beans, wheat, rice, sorghum and soybean); production volume data for each crop were first expressed as a proportion of the maximum reached as of 2007 and then, rescaled to 1/6 so that the sum of all the six crops would add up to a maximum of 1.



only way to elucidate which services are involved in a particular case (a pixel or a region in the country) is by looking up on the original maps (Figs. 8, 11 and 15). An alternate, more informative, manner to display combinations of values of up to three quantitative variables (= supply levels of ecosystem services in this case) is through a composite, false colour image. In colour composites, each variable is displayed as gradations of a single primary hue of light (red, green, or blue). Interpretation of these maps is straightforward, as the gradations of colour observed in the composite image are the result of combining the primary hues; these can go from dark (low values of all three variables) to white (high values of all three variables), areas in a "pure" colour (red, green, or blue) have high values of only one variable and low values of the other two, while white areas have high values of all variables, and black areas are low in all. Magenta is a mixture of red and blue, with low values of the green variable; yellow is high in green and red, with low values of blue; and cyan is high in green and blue, but low in red:



Fig.17b shows a composite, false colour map displaying the combined supply levels of the three ecosystem services considered. In this map, gradations of red represent levels of food crop provision, blue represents levels of surface water supply and green represents levels of Carbon sequestration. This map shows clearly (in reddish tones) that the major agricultural areas of the country, such as those along the coastal plain of Sonora-Sinaloa, supply high levels of food crops but do not contribute to Carbon sequestration and only very little to surface water supply. Ecosystems covering the central part of the Yucatan peninsula, as well as those in the Transverse Volcanic Axis and the Eastern Sierra Madre appear in bright green tones, revealing their high rates of Carbon sequestration but small contributions to the provision of surface water and even less to food crops. More importantly, the map shows clearly (in cyan tones) that the forests covering the eastern mountain ranges of Oaxaca, parts of the Southern Sierra Madre, as well as the northern, southern and Lacandon mountain ranges in the State of Chiapas have high rates of Carbon sequestration at the same time that supply large volumes of surface water. These ecosystems can clearly be considered as the country's "hotspots" for the valuable ecosystem services they supply; in addition, although not evaluated in this pilot study, the rich biodiversity that these ecosystems harbour is well known and has been widely documented.

Table 5 is an aggregated ecosystem service supply table; it shows the extent covered by each ecosystem type present in the country, as well as the volume (in physical terms) of the various services each of them supplies, for various points in time. Considering only on the three ecosystem services addressed in this pilot study, the evergreen tropical forests and coniferous forests still remaining in the country are the most "valuable" ecosystem types, as they are the ones that contribute the most to both, Carbon sequestration and surface water supply.

Grasslands and annual croplands deserve special mention. These two ecosystem types seemingly contribute huge volumes to the country's supply of surface water supply. However, this is due, first, simply to the large area they currently occupy in the country. Secondly, as used for this pilot study, the grassland category actually encompasses both natural grasslands and —in a large proportion— pasturelands. Most pasturelands currently existing in Mexico occupy land formerly covered by tropical rain forests which were cleared-cut in order to devote the land to livestock ranching and are, therefore, located in high-rainfall areas. More importantly, most of the water that runs-off in both pasturelands and



croplands is, in fact, consumptively used in place to support the maintenance of the pastureland as well as the growth of rain-fed crops.

Changes over time could only be examined for surface water supply and food crops provision; data are not yet available for analysing Carbon sequestration at different points in time. Results from the third sampling cycle (2015-2019) of the National Forest Inventory are due to be released sometime in 2020 and, with them, a new estimation of Carbon sequestration by the country's ecosystems could be produced using the same methodological approach developed for this pilot study.

As can be seen in Table 5, changes over time in the volume of surface water supplied by natural ecosystems are clearly related, first, to the reductions in extent that such ecosystems have been experiencing as they are converted to other land uses, and, secondly, to the conservation status of vegetation. By contrast, changes over time in food crops provision do not seem to follow any easily discernible trend and, thus, such variations seem to be rather related to either, fluctuations in weather or other environmental conditions (e.g., pests, diseases) that affect the crops yield or market conditions that affect the farmer's decisions to grow different crops in different amounts.

Fig. 17. Multiple ecosystem services supply: a) Top map, geographic distribution of a combined index of ecosystem services supply; the simple accumulation of the relative supply levels (expressed as a proportion of the maximum value reached as of 2007) of each of the three ecosystem services considered. B) Bottom map, a composite, false colour map displaying the combined supply levels of the three ecosystem services considered. Gradations of red represent levels of food crop supply, blue represents levels of surface water supply





and green represents levels of Carbon sequestration.



Table 5. Aggregated ecosystem service supply table showing the extent covered by each ecosystem type present in Mexico and the volume (in physical terms) of the various services each of them supplies, as of 2002, 2007, 2011 and 2014. Cells outlined in red colour identify the ecosystem types that most contribute to each ecosystem service. Data currently available do not yet allow estimating Carbon sequestration for different points in time. Food crop provision was only estimated for 2007, 2011 and 2014.

				2002	2007			2011			2014			
		Vegetation conservation	Extent (ha)	Surface runoff (m ³)	Extent (ha)	Total C sequestration (tm C/yr)	Surface runoff (m ³)	Food crops (tm)	Extent (ha)	Surface runoff (2011) (m3)	Food crops (tm)	Extent (ha)	Surface runoff (2011) (m3)	Food crops (tm)
	Coniformations	Primary	13,258,663	18,685,725,838	12,976,682	7,446,345	18,264,323,559	-	12,877,136	18,270,342,972	-	12,836,734	18,112,838,927	-
	Contrerus forest	Second growth	3,691,758	6,280,946,474	3,935,759	1,786,754	6,826,413,896	-	3,990,505	6,670,303,018	-	4,022,636	6,738,709,655	-
	Oakwoodland	Primary	10,982,234	10,484,159,320	10,734,926	3,555,156	10,095,086,595	-	10,773,540	10,221,491,556	-	10,728,812	10,141,077,834	-
	Oak wooulallu	Second growth	4,691,220	5,313,705,081	4,932,794	1,766,827	5,715,947,322	-	5,105,125	5,871,128,478	-	5,144,081	5,971,769,653	-
	Mountain cloud	Primary	1,293,826	5,054,490,570	1,329,202	1,909,159	5,157,911,906	-	1,333,162	5,151,572,638	-	1,324,138	5,122,862,067	-
	forest	Second growth	560,892	2,836,787,408	542,613	359,883	2,720,525,637	-	506,570	2,555,428,451	-	500,670	2,529,549,439	-
	Evergreen	Primary	7,960,042	24,119,948,330	8,176,855	8,793,739	24,984,413,659	-	8,178,011	24,360,554,135	-	8,012,055	23,824,395,114	-
	tropical forest	Second growth	2,580,805	11,915,822,163	2,124,222	625,616	9,695,892,667	-	1,980,876	9,256,499,329	-	1,947,368	9,062,052,164	-
	Semideciduous	Primary	2,909,930	4,011,185,774	3,005,848	3,683,920	4,038,933,216	-	2,870,860	3,909,712,015	-	2,805,166	3,818,365,948	-
	tropical forest	Second growth	1,868,193	3,646,160,953	1,452,850	1,240,476	2,915,615,066	-	1,191,833	2,483,319,300	-	1,197,010	1,863,651,851	-
	Deciduous	Primary	11,265,208	9,335,882,988	10,934,909	3,681,826	9,073,390,083	-	10,784,507	9,063,422,877	-	10,690,124	8,978,946,159	-
	tropical forest	Second growth	6,770,831	8,117,456,229	6,817,899	2,898,743	8,262,577,359	-	7,158,903	8,621,469,050	-	7,174,928	8,630,727,368	-
ms	Woody	Primary	18,758,155	4,896,549,103	18,543,874	921,651	4,843,140,274	-	18,376,092	4,731,759,712	-	18,313,906	4,703,847,968	-
	shrubland	Second growth	2,443,092	1,292,719,793	2,442,320	68,730	1,266,942,828	-	2,290,851	1,231,053,184	-	2,299,786	1,230,643,211	-
	Non-woody	Primary	34,582,582	5,052,235,157	34,174,486	259,938	4,982,297,516	-	33,698,212	4,892,445,217	-	33,648,956	4,884,224,543	-
	xerophytic	Second growth	2,531,008	579,973,971	2,662,849	- 100,410	633,163,887	-	2,979,625	707,862,009	-	3,084,096	720,226,104	-
	Other woody	Primany	259 825	109 682 351	266 124	10 951	118 125 726	_	240 418	132 898 980	_	236 297	126 748 526	_
	vegetation	i i i i i i i i i i i i i i i i i i i	233,023	105,002,351	200,124	10,551	110,120,720		240,410	132,050,500		230,237	120,740,520	
/ste	types	Second growth	168,598	128,158,144	161,936	- 4,679	115,583,648	-	178,556	123,508,186	-	180,954	128,446,801	-
cos)	Other non-	Primary	159,937	96,005,562	158,272	- 39,499	95,791,363	-	157,750	97,895,806	-	156,557	95,183,218	-
	vegetation	Second growth	ND	ND	ND	ND	ND	-	297	769,422	-	1,095	3,653,041	-
	Woody	Primary	1,089,376	1,848,012,282	1,098,562	755,801	1,870,270,936	-	1,101,095	1,868,640,646	-	1,098,584	1,863,651,851	-
	hydrophytic vegetation	Second growth	51,242	138,629,404	78,106	100,605	180,459,254	-	83,554	188,191,397	-	84,288	190,330,158	-
	Non-woody	Primary	1,480,294	4,556,896,703	1,436,658	25,805	4,441,017,532	-	1,463,473	4,597,211,581	-	1,451,393	4,547,033,636	-
	vegetation	Second growth	322	273,861	345	ND	286,459	-	5,402	449,633	-	5,402	449,633	-
	Grass	sland	31,620,377	53,007,194,936	31,156,391	8,439,289	53,587,257,357	-	30,745,606	52,892,167,951	-	30,885,225	53,159,530,358	-
	Other	lands	981,702	442,719,774	1,011,357	ND	477,421,669	-	1,045,404	486,015,017	-	1,055,376	502,643,578	-
	Forest pl	antation	32,114	104,377,155	37,354	37,354	124,826,629	-	65,257	230,966,543	-	75,100	271,784,644	-
	Aquac	ulture	70,081	10,843,854	94,501	-	18,644,603	-	113,054	21,014,105	-	120,692	22,325,441	-
	Human se	ttlements	1,267,254	1,650,446,891	1,607,122	-	2,153,237,276	-	2,143,929	3,108,702,351	-	2,183,804	3,195,230,754	-
	Perenni	ial crops	1,656,757	7,216,668,542	1,716,581	ND	7,382,731,253	ND	1,798,681	7,898,703,869	ND	1,859,572	8,198,994,516	ND
	Annua	l crops	29,421,034	36,806,697,273	30,738,869	-	38,332,534,200	Maize: 21,777,449 Beans: 993,952 Wheat: 3,515,392 Rice: 294,697 Sorgum: 6,202,920 Soybean: 88,371	31,041,495	39,892,094,006	Maize: 15,873,783 Beans: 567,779 Wheat: 3,627,510 Rice: 173,460 Sorgum: 6,429,311 Soybean: 205,233	31,165,524	40,518,385,822	Maize: 20,710,883 Beans: 1,273,957 Wheat: 3,669,813 Rice: 232,158 Sorgum: 8,394,056 Soybean: 387,366

9. Concluding remarks

Three major objectives are being pursued by the pilot studies carried out under the NCAVES-Mexico project:

- a) Test whether the SEEA-EEA approach for measuring and monitoring ecosystem services flows can be applied in Mexico;
- b) examine whether the results from the pilot accounts would yield information relevant to decision-making about natural resource management in Mexico; and
- c) identify how the experience of modelling ecosystem services and compiling the accounts for Mexico's conditions can contribute to the on-going development of the SEEA Experimental Ecosystem Accounting framework.
- a) First, we aimed to test whether —and to what extent— the SEEA-EEA approach can be applied in Mexico. In practice, this entailed examining whether —and how— multiple ES can be spatially modelled and analysed —using existing data and for the highly heterogeneous, vast expanse of Mexico's landscape— in a way suitable for ecosystem accounting as *per* the SEEA-EEA framework.



Our results showed that country-wide spatial modelling for ecosystem accounting purposes in line with the SEEA EEA is indeed feasible at least for a few selected ecosystem services. We developed spatial biophysical models of three key ecosystem services, *viz.*, Carbon sequestration, surface water supply and food crop production, plus a fourth one (coastal protection by mangrove ecosystems) that is still in process. However, a number of practical/technical issues were found and had to be addressed in order to do so. Such challenges were related to data availability, data harmonization and integration, biophysical modelling, and the strict requirements of the SEEA-EEA framework.

A first, key criterion for selecting the ecosystem services to be addressed in this pilot study was the availability of existent data with the appropriate qualities. We thought that sufficient, goodquality data would be indeed available in Mexico for examining the four ecosystem services selected. Actual implementation of the studies proved this to be a somewhat optimistic view.

First, the basis for compiling the ecosystem extension accounts —and, indeed, for building all the models used— was the four latest editions of INEGI's Vegetation and Land-use charts. INEGI's charts are, arguably, the best data sets currently available on the geographic distribution of Mexico's terrestrial ecosystems. However, their relatively coarse spatial resolution (minimum mappable area: 25ha) and low update frequency (every 4-5 years) are perhaps insufficient for a large, environmentally heterogeneous country with rapid vegetation and land-use change dynamics such as Mexico and limit the usefulness of the results obtained. Such limitations might not be noticeable/significant when looking at total, country-wide figures for a given point in time but might be significant when trying to disaggregate the results at State- or municipal-level, and when trying to examine changes over time. More efforts are needed to produce higher quality, wall-to-wall charts of Mexico's terrestrial ecosystems; that is, charts yielding a higher spatial and temporal resolution at the same time that maintaining a detailed semantic resolution.

The different editions of INEGI's Vegetation and Land-use chart were produced at different dates and using different data sources. Because of this, when the charts are examined comparatively, real variations/changes that have occurred in the terrain over time as well as artifacts derived from the different origin and pre-processing of the source data show up, leading to anomalous results. For example, they produce minor (but not insignificant) variations in the total extent of the country (and, therefore, of some of the ecosystems contained therein) between different editions of the chart. To address this particular issue, INEGI's experts re-examined the charts in detail and produced, for these pilot studies, an *ad hoc* revised version in which the coastline, the limits between marine and coastal waters, and the country's boundaries were consistently identified and traced. However, some other inconsistencies still remain that become evident when the charts are overlaid to examine changes over time (i.e., from one edition to another) and spurious, impossible changes in vegetation or land use appear. Such issues are common in this sort of studies and are generated by combining spatial data from different sources that have been subject to different processing procedures (see *e.g.*, Crossman *et al.*,2013; Martínez-Harms and Balvanera, 2012; Nemec and Raudsepp-Hearne, 2013; Keith *et al.*, 2017), and introduce uncertainty in the results obtained. At the moment there is no practicable way to really resolve all those inconsistencies.

Secondly, the major issue faced when trying to map and assess the condition of Mexico's ecosystems and its changes over time was data availability. As described in the report on *Assessing ecosystem extent and condition in Mexico*, for these pilot studies we had to rely mostly on statistical and geographical data that are regularly collected by various government agencies as part of their operation and that are relevant and —after suitable reformatting, adaptation, or reformulation—potentially usable for these purposes. This way, seven variables/indicators/indices of ecosystem



condition were initially identified. However, upon closer examination, fully suitable data were available for only two of those, namely, the conservation status of vegetation, and the Human Footprint Index. Data currently unavailable for other relevant variables/indices (*e.g.*, Ecosystem Integrity Index and soil erosion) might be produced or made available in the near future. However, suitable data on other key variables such as soil organic carbon content and biodiversity will be more difficult to obtain. Vast amounts of data relevant for examining ecosystem condition and its changes over time are regularly collected in Mexico through soil surveys, the national forest inventory, the national system of information on biodiversity, and other national programmes. However, being compiled for purposes other than ecosystem accounting, those data sets do not readily meet the requirements necessary to be used for this purpose. More efforts are needed to collect/compile country-wide, spatially explicit, moderate resolution, multi-date data on biophysical variables/indicators of ecosystem condition, using existent and traditional data sources (*e.g.*, field monitoring), data from other sources (*e.g.*, remote sensing), and suitable modelling tools (*e.g.* S-World).

The lack of suitable data for assessing the condition of Mexico's ecosystems has been further compounded by the updated guidelines from the SEEA-EEA revision process (UN, 2018). Among the early results from such revision, a set of 12 selection criteria that ecosystem condition indicators should comply (under the SEEA-EEA framework) was formulated (Keith et al., 2019; Czúcz et al., 2019). As discussed in the report on Assessing ecosystem extent and condition in Mexico, while the selection criteria would help ensure the logical consistency of the SEEA-EEA ecosystem condition accounts, they also impose strict requirements that might be difficult to meet in not-so data-rich countries thus making the application of the SEEA-EEA framework even more challenging. In our case, the only two variables/indices of ecosystem condition for which suitable data were available (viz., conservation status of vegetation and the Human Footprint Index) fail to meet all the selection criteria. The conservation status of vegetation is clearly a highly informative variable for which, however, a suitable reference level cannot be easily identified/set, thus violating the SEEA-EEA's normativity criterion. Similarly, pressure indicators such as the Human Footprint Index violate the framework consistency criterion and their use as indicators of ecosystem condition is explicitly discouraged. This is, however, the best information related to ecosystem condition that is currently available in Mexico.

Third, spatially-explicit, biophysical models had to be developed for each of the ecosystem services selected, and the modelling outcomes then used to construct ecosystem accounting tables listing (as *per* the SEEA-EEA guidelines) the estimated contributions of the different ecosystem types to ecosystem service flows. Over 150 local-level studies of ecosystem services valuation have been conducted in Mexico over the last 25 yr and several of them have addressed these same ecosystem services. The information compiled and the results produced by those studies on the volume, flow and value of ecosystem services would have been the natural starting point for the pilot projects of the NCAVES-Mexico project. Unfortunately, as discussed at length in *Mexico's country assessment* report, such studies are affected by technical issues that prevent their results from being more generally useful and able to be extrapolated to the entire country. For this reason, for these pilot studies I also had to rely, primarily, on data collected by various government agencies as part of their operation. While such data are fully relevant and potentially useable for these purposes, having been collected for purposes other than ecosystem services accounting, they had to be first reformatted, adapted, reformulated, used as inputs for biophysical models/analyses, and merged with other data, prior to their use for estimating ecosystem services stocks and flows, etc. A major issue was that some of those data are not spatially explicit (or only partially spatial), and the models were then built by combining spatial and non-spatial information. Combining different data types, with different degrees of spatial explicitness and spatial variation, necessarily introduced some inconsistencies and


uncertainties in the results obtained which cannot be easily quantified. Moreover, not all the data necessary for building the models were available from those sources.

For example, the georeferenced field data from the National Forest Inventory —recognized as the very best forest C dataset available in Mexico— were the basis for estimating C density and between-dates differences in C stocks. However, in their original intended use, those data are aggregated and processed to produce country-wide estimates per ecosystem type, in a tabular format. To use them for these pilot studies, the NFI field data had first to be combined with other, separate data sets, viz., INEGI's Vegetation and Land-use charts and the CEC's map of North American Terrestrial Ecoregions, to obtain regionalized estimates of C density (per ecosystem type per ecoregion) which could then be mapped over the whole country. The coarse, and different, horizontal resolution of the two maps used for spatializing the C density estimates necessarily introduces limitations on the validity of the results obtained. Such limitations might not be noticeable/significant when looking at total, country-wide figures but will certainly be significant when trying to disaggregate the results at the State- or municipal-level. In addition, we were unable to report on the C stocks and emissions from soil organic carbon due to the lack of suitable data (see discussion in Annex A). This is an important gap in our results as at least some soil types might contain a significant stock of C and be a potential source of C emissions. Thus, although this analysis was based on the best datasets available for the country, these are not perfect and led to some inconsistencies and uncertainties in the results obtained. It would be desirable to carry out an independent uncertainty analysis of the results but, to date, no other, more accurate data set is currently available.

Similar issues were faced in the food crop production study. The core data set used for this analysis was the detailed, country-wide, official statistics of crop production that the Ministry of Agriculture compiles and publishes annually. These data are only partially spatial in nature and delivered in tabular form, aggregated at the municipality level; being derived from administrative records, they cannot be disaggregated any further. In order to use these data for the pilot study, I had to first merge them with INEGI's Vegetation and Land-use charts to approximately map the production figures onto the agricultural parcels present in each municipality (as delineated in INEGI's charts) of the country. While the model's outputs are consistent with SIAP's data, and successfully recover the overall distribution pattern of food crop production (as for maize, bean, wheat, rice, sorghum and soybean) across the country, the empirical approach utilized to spatialize the data necessarily introduces inaccuracies and uncertainties. First, as INEGI's charts use a minimum mappable of 25ha, all production data points coming from smaller areas are omitted in the resulting maps. More importantly, all the model's numerical outputs (viz., production figures for a given cropland area or polygon) are, in reality, imputed values rather than actual measurements or estimates. This limits the validity and usefulness of the results. For instance, the model's results cannot be validly correlated with ecosystem condition indicators, distribution of protected areas, etc. Obviously, the individual figures (*i.e.*, for a given polygon) that the models provide can be used only in relative terms but their absolute values should be interpreted with caution. These limitations/uncertainties will affect, to some extent, the use of the model's outputs for valuing food crop provision in monetary terms. Specific efforts to monitor agricultural production in a spatiallyexplicit manner that complement Mexico's production statistics are needed.

Even more serious limitations were faced in the study on surface water supply. The lack of detailed, country-wide, time-series data sets precluded the use of complex, process-oriented, but data-demanding hydrological models. Instead, a simple but spatially-explicit, empirical model that utilizes readily available data was utilized. While the model was able to capture reasonably well the overall pattern of surface runoff across the country, both at the pixel- and the drainage basin level, the model's outputs underestimate surface water supply in most basins and, in some cases, fail to



capture the interactions between infiltration and surface runoff. In particular, the model greatly overestimates surface water supply for the unique physical conditions of the Yucatan peninsula. Thus, the model's numerical outputs can only be used in relative terms but their absolute values should be interpreted with caution. These limitations/uncertainties will seriously limit the use of the model's outputs for valuing surface water supply in monetary terms. Therefore, more detailed, process-based models are necessary to properly describe and value the relationship between ecosystems and surface water supply in Mexico; however, the much more detailed data that they demand are not yet available.

Despite the above mentioned shortcomings, the rather simplistic, but spatially-explicit, models developed for this pilot study —utilizing data that are readily available— do take into account the spatial distribution, extent and condition of Mexico's ecosystems and land-use types; their relationship with key environmental variables (*e.g.*, soils, rainfall, geology, etc.); and are thus able to capture the overall pattern of C storage and sequestration, surface water runoff and food crop production across the country and how these change over time. They are also able to show how the supply of these ecosystem services varies in relation to environmental conditions, and with the extent and condition of ecosystems and land-use patterns. Finally, the models can also depict how the different ecosystem types contribute to each of these services and identify the specific ecosystems supplying these services in a given region. In summary, the models hereby developed do provide useful information and, despite their limitations, can be adopted as an initial stage (tier 1) in the design of suitable methods for the valuation of key services provided by Mexico's ecosystems. As additional, better data become available, these models can be refined and their limitations overcome, or other more complex, data-demanding, models can be implemented.

As discussed above (in the section **Mapping and valuing ecosystem services in Mexico**), in its ambition for integrating ecosystem services accounts into national economic accounts, the SEEA-EEA framework imposes strict conceptual and methodological requirements on how ecosystem services should be valued in physical and monetary terms. One of those is its narrow definition of ecosystem services and the sharp distinction between ecosystem services and benefits: For each good or service derived from ecosystems, the actual contribution that ecosystems make should be disentangled from what is generated by/through human inputs and interventions. In practice, however, this might be very difficult to achieve —at least in some cases — given current data and knowledge limitations.

For example, the availability of fresh water is determined, to a large extent, by the interplay of physical processes; ecosystems only modify the amount and properties of water moving/passing through them. Therefore, in order to properly evaluate the ecosystem service of provisioning fresh water for anthropogenic uses, those two components should be disentangled, but this is extremely difficult to achieve in practice. In fact, our study on surface water supply did not make this crucial distinction. A similar issue was faced in the food crop production study. Agricultural production is determined by a variety of human inputs plus a range of ecosystem services. For accounting purposes under the SEEA-EEA framework, the natural processes that enable and contribute to agricultural production should be identified and quantified to properly determine the ecosystem contribution. In practice, however, disentangling these two components is also extremely difficult. In our study we simply used the volume of crop before harvest as the physical measure of the final ecosystem service, fully aware that this is not the best measure of this ecosystem service.

These last issues are also relevant for the next stage of the NCAVES-Mexico project in which the results presented herein are to be used for valuing these ecosystem services in monetary terms. As *per* the SEEA-EEA framework, ecosystem services should be valued in terms of their exchange



value so that the results are logically consistent with and can be directly integrated into national economic accounts. The unit resource rent approach is one of the methods most recommended for valuing provision services (such as water supply and crop provision) in a manner consistent with the SEEA-EEA requirements. However, as discussed above, this method might be difficult to apply in Mexico due to the particular conditions prevailing in its agricultural sector (*e.g.*, subsidies, access to natural resources at no or very low cost, etc.) which might yield zero or even negative values for these ecosystem services.

While the requirement of valuing ecosystem services in terms of their exchange value would ensure the logical consistency of the ecosystem services valuation accounts with national economic accounts, it might be a difficult to achieve goal in not-so data-rich countries thus making the application of the SEEA-EEA framework even more challenging. Perhaps a looser approach to the monetary valuation of ecosystem services might be useful in those cases; even if the results obtained cannot be readily incorporated into the national accounts, they might, nevertheless, help appreciate better (perhaps only in relative terms) the value/importance of ecosystems and the services they provide.

b) Secondly, we aim to examine whether the results from the pilot accounts would yield information relevant to decision-making about natural resource management in Mexico, and how the results from the pilot accounts may be used by government institutions involved in natural resource management.

The results presented herein only comprise the evaluation of ecosystem services in physical terms; in the on-going and final stage of the NCAVES-Mexico pilot study these results are being used for valuing these ecosystem services in monetary terms. Thus, this second objective would be accomplished once the two sets of results are completed and submitted to the consideration of stakeholders. Nevertheless, the results obtained so far have been presented to and discussed with officials from relevant government agencies. Such exchanges have evoked reactions and raised interest as to the usefulness of these partial results for supporting the implementation of environmental policies and programmes in Mexico, as follows:

• <u>Data gaps</u>.- As detailed in the previous sections, a number of data gaps and limitations have surfaced or become more evident during the development of these studies. Discussion of those issues with relevant parties has already elicited some responses and actions to try to address them.

As mentioned above (p.68), INEGI's experts re-examined the Vegetation and Land-use charts in detail to produce an *ad hoc* revised version in which the coastline, the limits between marine and coastal waters, and the country's boundaries were consistently identified and traced in order to reduce the between-charts inconsistencies in the total extent of the country. Also, in order to ameliorate other inconsistencies that appear when the charts are overlaid to examine changes over time and spurious, impossible changes in vegetation or land use appear, INEGI is entertaining the possibility of producing a (250m resolution) raster version of these charts. Preliminary tests of this idea have shown promising results.

An important shortcoming of INEGI's Vegetation and Land-use charts is their relatively low spatial and temporal resolution. INEGI is currently working on improving its methods and procedures and the next iteration (series 7) of this chart, due to be published in 2021, might have an improved horizontal resolution. As these charts are the basis for the ecosystem extent account and for the construction of the ecosystem services models hereby utilized, these advances will lead to significant improvements in the next iteration of the ecosystem services study.



Moreover, INEGI is currently conducting efforts to use advanced statistical models as well as biophysical models (*e.g.* the S-World approach) to exploit the vast set of soil data that they (together with CONAFOR) have been compiling in the field over the last 50 yr in order to map soil organic C for different points in time. This would fill a major data gap in both, the ecosystem condition accounts, and the evaluation of C storage and sequestration; aside from the ecosystem services project, soil organic C data are also a key input for the national GHG emissions inventory.

Upon the specific request of the NCAVES-Mexico project, SEMARNAT decided to produce a second edition of its Human Footprint Index (with data for 2014-2015) for the compilation of ecosystem condition accounts. Equally important, during the *interim* technical exchanges between INEGI and SEMARNAT experts, possible improvements to the technical specifications of the index to make it more responsive to different levels of transformation by human activities, as well as the possibility of including additional information on relevant human activities were identified. If implemented, a new, improved version of the Human Footprint Index would soon be produced. Aside from the ecosystem services project, a robust, synthetic measure of human pressures on ecosystems would be most useful for state of the environment reporting.

A simplistic, empirical approach had to be used to spatialize the statistical production data compiled by the Ministry of Agriculture. While the resulting models provide a credible picture of the geographic distribution of agricultural production in Mexico, they bear important shortcomings. The Ministry of Agriculture is currently undertaking two different projects aimed at mapping agricultural land in Mexico, although they have not yet attempted to link the crop production data with these cartographic products. Combining the analytic approach developed hereby with these new cartographic products might likely improve the evaluation of food crop provision ecosystem service, as well as setting the foundations for the construction of a robust means to examine, in a fully spatially-explicit manner, the location of and changes over time in agricultural production in Mexico, a key input for many aspects of agricultural planning and policy implementation.

Finally, the simple empirical approach hereby developed to spatialize the C storage data obtained by CONAFOR provides important, spatially-explicit, additional information that helps to identify the zones where C sequestration and C stocks are the highest, as well as the ecosystem types that are most valuable in this regard. Given the potential relevance and applications of these results, it would be important to conduct an independent validation/uncertainty analysis of them. No other, more accurate data set is currently available; however, a good opportunity will be provided by the field data from the third sampling cycle of the National Forest Inventory, which is to be completed in 2020. The new data set will allow not only validating and, if necessary, adjusting the results of the C sequestration analysis but, equally important, will allow estimating C sequestration rates for a subsequent time period (2009-2014 to 2015-2019) and thus examine changes over time in this ecosystem service.

In summary, through the NCAVES-Mexico pilot project a number of initiatives have been identified —or even elicited— that are going to produce formerly unavailable or better information that not only will improve significantly the next iteration of this ecosystem services study but will also be useful to support environmental planning and the implementation of various environmental programmes in Mexico.

• <u>Potential usefulness and policy-relevance of the results.-</u> The efforts to map and value (in physical terms) C sequestration, surface water supply and food crop production made in this project, have been found interesting and their results potentially useful —even in the absence of the monetary valuation— to support or improve the implementation of various governmental environmental programmes; they can also complement similar or related efforts being carried out by government agencies such as the Ministry of Agriculture.



Despite their limitations, the spatial modelling approaches developed in this study can be used as an initial step in the design of suitable methods for the valuation of ecosystem services in Mexico. These methods could thus help SEMARNAT and CONAFOR to fulfil their mandate to "... define the methodologies for the valuation of environmental goods and services provided by forest ecosystems..." and "... propose the valuation of environmental goods and services of forest ecosystems, according to the methodologies defined by [SEMARNAT]...". As additional, better data become available, these models can be further refined (tier 2), or better, more sophisticated models can be implemented.

Even though the models' outputs were not accurate enough, they constitute additional, spatially-explicit information that can be useful to support or improve the implementation of key environmental management policies, instruments and government programs such as the Payment for Hydrologic Environmental Services and the watershed restoration programmes (in charge of CONAFOR). Equally important, those results would be useful for INECC as a baseline for assessing the likely impacts of climate change on the supply of critical ecosystem services in different parts of the country.

Results obtained in the C sequestration study would also be useful for INECC as they convincingly document the contribution that Mexican forests make to Mexico's mitigation goals pledged in its Nationally Determined Contribution. The C sequestration and C stocks maps allow identifying those parts of the country or those ecosystems that are most valuable in this regard so that they can be prioritized in conservation or incentives programmes implemented by agencies such as CONAFOR and CONANP. If replicated consistently over time, these analyses could then be used by INECC to monitor progress towards Mexico's climate change mitigation obligations, as well as by SEMARNAT to track the progress and effectiveness of forest conservation and management programmes.

The improvements made by INEGI experts to the assignment tables of the indirect method in NOM-011 could be incorporated by CONAGUA into an updated, improved version of this national standard. Moreover, the improved method might be directly used by CONAGUA to produce refined estimates of water availability in those watersheds for which sufficient field data are not available. Albeit with some limitations, outputs from the surface water supply models can help identify the ecosystems that contribute the most to the supply of water for anthropogenic uses in a given region, and its beneficiaries. The key water-supplying zones or ecosystems thus identified can then: a) be prioritized in conservation or improved land-use programmes; b) be "flagged" to prevent their being negatively affected by development or infrastructure projects; c) help reveal potential trade-offs between alternative land management decisions; etc. A particular example of this use was pointed out by various experts during the exchanges: Correlating the spatial models developed for food crop production with those for water surface supply would help to link, in a spatially-explicit manner, agricultural production with the ecosystems that provide them with essential services such as water supply. This information could then be used for the design of local schemes through which the direct beneficiaries provide material incentives to the ecosystem owners to protect the ecosystems that supply the water they benefit from.

In summary, as recognized by various experts from relevant government agencies, despite their limitations, the results obtained in the valuation in physical terms of select ecosystem services provide, by themselves (*i.e.*, even in the absence of the valuation in monetary terms), useful/valuable information to support environmental planning and improve the implementation of environmental programmes and policies in Mexico.

c) Third, we aimed to identify how the experience of modelling ecosystem services and compiling the accounts for Mexico's conditions can contribute to the on-going development of the SEEA Experimental Ecosystem Accounting framework. This objective would only be accomplished once the results from the valuation in monetary terms are also completed and submitted to the consideration



of UNSD and the bodies (e.g., the UN-CEEA) in charge of developing the UN SEEA-EEA framework. Nevertheless, the work carried out so far to value ecosystem services in physical terms has highlighted two points that might be worth pointing out hereby.

First, it is likely that the same (or similar) challenges that we faced during the implementation of the NCAVES pilot project in Mexico might also happen in other countries when trying to compile country-wide evaluations of key ecosystem services as *per* the SEEA-EEA guidelines. Thus, the criteria adopted, the decisions made, the methodological approaches developed, and the lessons learned during the implementation of the pilot studies in Mexico might constitute useful reference for those other countries. I would therefore suggest that these findings be shared and discussed with relevant experts from the other countries participating in the NCAVES project as well as those participating in the closely related WAVES project.

Second, one particular point that has surfaced repeatedly over the course of the implementation of this pilot study is the lack or scarcity in Mexico of relevant, robust, high-quality, country-wide, multi-date data on ecosystem services in general. We found even fewer data that meet the strict requirements of the SEEA-EEA methodological framework —which have been further compounded recently by the results from the SEEA-EEA revision process. While these strict data requirements would ensure the logical consistency of the ecosystem accounts, they made the application of the SEEA-EEA framework challenging and, as this report shows, in some cases we were able to produce only partial results.

Although the economic valuation component of the NCAVES-Mexico project is still in process, there is evidence that suggests that the SEEA-EEA's strict requirements for the economic valuation of ecosystem services might be difficult to fulfil at least in some cases. This is not only due to the unavailability of sufficiently detailed information but also to the presence of subsidies, zero rate fees, etc. in many agriculture-related services in Mexico.

From my experience, it is likely that similar or even worse limitations occur in other not-so data-rich countries. In those cases, perhaps a gradual approach to the valuation of ecosystems and ecosystem services might be more feasible to achieve. Ecosystem services valuation studies could initially focus only on producing robust estimates of the services supply and use in physical terms. The initial stage might then be followed by the economic valuation of those ecosystem services using the most rigorous approach that is feasible. Even if the results cannot be readily integrated into the national accounts, they might, nevertheless, help appreciate better (perhaps only in relative terms) the value/importance of ecosystems and the services they provide. The final stage of the process would aim to fully implement the SEEA-EEA conceptual and methodological framework.



APPENDIX





Figure A.1a. Changes in the extent of Mexico's ecosystems over time: Natural ecosystems. PA: Protected areas.





Figure A.1b. Changes in the extent of Mexico's ecosystems over time: Anthropogenic ecosystems. PA: Protected areas.



NATURAL CAPITAL ACCOUNTING AND VALUATION OF ECOSYSTEM SERVICES PROJECT

Pilot testing the SEEA-EEA framework in Mexico ANNEX A: Mapping and assessing ecosystem extent and condition



1. Preamble

One product of the *Natural Capital Accounting and Valuation of Ecosystem Services in Mexico Project* shall be a compilation of selected ecosystem services accounts in physical and —whenever possible— monetary terms. Based on the country's policy priorities, data availability, and on their sensitivity to environmental changes and socio-economic importance (see UN, 2014), the following four ecosystem services were chosen for pilot testing the SEEA-EEA framework in Mexico both, at the Stateand the countrywide level:

- Carbon capture and storage services
- Surface water supply services
- Food crop production services provided by agroecosystems
- Coastal protection by mangrove ecosystems

Given the still experimental nature of the SEEA-EEA framework, its strict methodological requirements, as well as its very recent —and, therefore, limited— application in real-world cases, it is not surprising that challenges are found in various stages of the implementation process. In this document I discuss, in particular, the issues faced when trying to map and assess the extent and condition of Mexico's ecosystems, the rationale underlying the decisions made, as well as the gaps that need to be filled in order to make further progress in ecosystem services accounting in Mexico.

Work referred to in this and other accompanying reports was carried out during 2018 and early 2019, during which period a revision process of the SEEA-EEA conceptual and methodological framework was also being conducted (UN, 2018). Early results from the revision process were released very recently (May-June, 2019) and, while they do provide further guidance and clarification on these topics, timing precluded incorporating these new developments into the Mexico pilot studies. Nevertheless, in this document I try to frame —as much as feasible— the issues found and the decisions made in relation to the results from the revision process.

2. Ecosystem classification

The first step in the SEEA-EEA framework involves delineating the ecosystems (*ecosystem assets*, as per UN, 2017) present in the study area (*ecosystem accounting area*, as per UN, 2017). The rationale is that for ecosystem accounting purposes, the biophysical environment can/should be partitioned into a set of contiguous, mutually exclusive units, each covered by a specific ecosystem that supplies ecosystem services and about which relevant information is to be collected as the basis for accounting.

In principle, a range of characteristics can be used to delineate the ecosystems present in the study area, including ecological (*e.g.*, vegetation type, soil type, hydrology, land management, etc.) and non-ecological (*e.g.*, land use, ownership, site features, etc.) features. Accordingly, many different approaches and systems for classifying ecosystems have been proposed and are in use worldwide. Ecosystem classification systems commonly differ from each other in the characteristics or attributes of ecosystems that they use (or emphasize) as classification criteria, depending on the scope (local, national, regional or global) and specific purpose for which the classification is to be used.

However, the UN-SEEA ecosystem accounting concept demands ecosystem classifications that are suitable for statistical analysis and accounting and —given the UN's essence— that also allow for the results to be compared across countries. The SEEA-EEA document (UN, 2014) and its technical recommendations (UN, 2017) focused very much on terrestrial ecosystems and suggested a simple land-cover classification (based on version 3 of the UN-FAO's Land Cover Classification System) as a starting point for classifying ecosystems. Early experiences with the SEEA-EEA framework have evidenced some shortcomings in such classification (Bogaart *et al.*, 2019), including the lack of detail with regard to



marine ecosystems. Thus, although the general approach for classifying ecosystems in the SEEA-EEA context is now relatively well established, there are still several important issues that need to be addressed; definite approaches and methods for delineating ecosystem assets for ecosystem accounting purposes are still under development (UN, 2017; Bogaart *et al.*, 2019).

It has been now recognized that, for ecosystem accounting purposes, land cover is, by itself, insufficient for properly delineating ecosystem assets and a wider range of ecological (biotic and abiotic) and non-ecological characteristics needs to be considered (Bogaart *et al.*, 2019). Ecological characteristics are usually chosen as the key criteria for delineating ecosystem assets; for terrestrial ecosystems these include (Bogaart *et al.*, 2019):

- Biotic characteristics.- Vegetation structure, type, and species composition; fauna; ecological processes, etc.
- Abiotic characteristics.- Climate, topography and geomorphology, lithology and soil, hydrology, etc.

The specific characteristics to use in a particular case should be chosen based on their relevance for the ecosystems involved and the availability of data. Habitat/biotope and vegetation are usually the most suitable basis for delineating and classifying terrestrial ecosystems, based on ecological criteria (Bogaart *et al.*, 2019). If available, a country-specific classification system would also be strongly preferable as this will neatly reflect the local conditions; however, a highly specialized classification might limit the possibilities of comparing the results with those from other countries (UN, 2017).

In Mexico, several, progressively more refined, systems for describing and characterizing the country's vegetation have been formulated over the course of the last 60 years (Sánchez-Colón *et al.*, 2009). The first country-wide classification and description of Mexico's vegetation types was proposed by A.S. Leopold in 1950. This was followed, several years later, by Miranda and Hernández-Xolocotzi (1963) who published *Los tipos de vegetación de México y su clasificación* (Classification of Mexico's vegetation types). Several studies were completed in the 1970's, including the *Mapa y descripción de los tipos de vegetación de la República Mexicana* (A map and description of the vegetation types of the Mexican Republic) by Flores Mata *et al.* (1971), the *Tipos de vegetación de México* (Vegetation types of Mexico) by González Quintero (1974) and *Vegetación de México* (Mexico's vegetation) by Rzedowski (1978). These seminal studies provided, for the first time, an overall picture of the country's vegetation and the different plant communities that it comprises, their main and dominant species, ecological and geographical distribution and even provided information on the anthropic pressures affecting them. In addition, those first studies set up a general framework for the further description and analysis of the plant communities, vegetation and terrestrial ecosystems of Mexico.

As part of their long-standing work to periodically map the country's vegetation and land cover, the National Institute for Statistics and Geography (INEGI) has formulated a comprehensive classification system of the vegetation and land-use types present in Mexico. INEGI's classification system is based on those proposed by Miranda and Hernández-Xolocotzi (1963) and Rzedowsky (1978), and is nowadays the most comprehensive, highly-detailed, country-wide level classification of the main types of natural vegetation and land use occurring in the country. The system has been progressively refined over the last 40 years and is now consistently used as a framework for describing Mexican terrestrial ecosystems in all environmental planning and management instruments (*e.g.*, National Forest and Soil Inventory, land-use planning schemes, environmental impact assessment of development projects, etc.) and national reports



and indicators (Mexico's *State of the Environment Report, National System of Environmental Indicators*, etc.) produced in Mexico, as well as in country reports to multilateral organizations and conventions (*e.g.*, FAO-*Forest Resource Assessment*, National Communications to the United Nations Framework Convention on Climate Change, National Reports before the Convention on Biological Diversity, etc.). In its latest version, the INEGI classification encompasses 12 plant formations (including conifer forests, oak forests, mountain cloud forest, xerophytic shrubland, natural grassland, evergreen tropical forest, deciduous tropical forest, semi-deciduous tropical forest, thorny tropical forest, hydrophytic vegetation, and others) subdivided into 59 vegetation types (differentiated based on physiognomy, dominant species and environmental conditions) plus 24 land-use classes (including agroecosystems, human settlements and water bodies); a detailed description of INEGI classification system is provided elsewhere (INEGI, 2017).

More recently, González Medrano (2003) made an effort to synthesize and unify the different classification systems of Mexico's vegetation. Velázquez *et al.* (2016) proposed a standardized hierarchical classification system for Mexico's vegetation. However, these more recently proposed classification systems have not been widely adopted or used so far.

Based on the above, and taking into account the general considerations described in section 3, the ecosystem classification chosen for the pilot tests of the NCAVES-Mexico project was based, first, on INEGI's vegetation and land use classification system (INEGI, 2014, 2017). The level of detail of INEGI's classification system, however, surpasses the data available on ecosystem services and, for this reason, a simplified version of the INEGI classification was used instead. Specifically, I chose the aggregated version that the National Forestry Commission (CONAFOR) utilizes as framework for compiling the national greenhouse gas emissions inventory from the land use, land use change and forestry sectors. In such simplified scheme, the 59 vegetation types and 24 land use classes of the original INEGI classification are aggregated into 14 vegetation classes, five land-use classes (annual or permanent crops, aquaculture, forest plantations, human settlements) and ancillary classes (water bodies). Table A.1 shows the correspondence between INEGI's full classification and the aggregated classes included in CONAFOR's simplified version.

Table A.1. Correspondence between the vegetation and land-use classes encompassed in INEGI's full classification and the
aggregated classes included in CONAFOR's simplified classification; their approximate correspondence with IUCN's Global
Ecosystem Typology is also shown. Cells shaded in pink colour denote land-use or ancillary classes; cells shaded in green
colour denote natural vegetation classes.

INEGI class	CONAFOR class	IUCN Global Ecosystem Typology
Aquaculture	Aquaculture	F3 Artificial wetlands
Water bodies	Water bodies	F1 Rivers and streams/F2 Lakes/F3 Artificial wetlands
Annual crops dependent on residual moisture		
Annual + perennial crops dependent on residual moisture		
Annual + semi-permanent crops dependent on residual moisture		
Annual, irrigated crops		
Perennial, irrigated crops		
Annual +semi-permanent, irrigated crops	Annual crops	T7 1 Cronlands
Semi-permanent, irrigated crops		
Semi-permanent + perennial, irrigated crops		
Annual, rain-fed crops		
Annual + perennial rain-fed crops		
Annual + semi-permanent rain-fed crops		
Semi-permanent rain-fed crops		
Semi-permanent + perennial rain-fed crops		
Perennial crops dependent on residual moisture	Perennial crops	



Semi-permanent crops dependent on residual moisture				
Semi-permanent + perennial crops dependent on				
residual moisture				
Perennial, irrigated crops				
Perennial, rain-fed crops				
Urban/constructions	11	TT Alluhan and informations lands		
Urban zones	Human settlements	I 7.4 Urban and infrastructure lands		
Forest plantation	Forest plantation	T7.3 Plantations		
Fir forest				
Cypress forest				
Juniper forest		T2.1 Developed montane readle locued		
Pine forest	Coniferous forest	forest and woodlands		
Pine-oak mixed forest				
Douglas fir forest				
Coniferous shrubland				
Oak forest	Ook foract	T2.2 Temperate deciduous forests and		
Oak-pine mixed forest	Oak lorest	shrubland		
Mountain cloud forest	Mountain cloud forest	T2.4 Warm temperate rainforests		
Tropical mezquital				
Subtropical shrubland	Desidence transferd			
Low-stature, deciduous tropical forest	Deciduous tropical	T1 2 Teasier l/a blassier late francts and		
Low-stature, deciduous, thorny tropical forest	Torest	11.2 Tropical/subtropical dry forests and		
Mid-stature, deciduous tropical forest	1	scrubs		
Low-stature, semi-deciduous tropical forest	Semi-deciduous			
Mid-stature, semi-deciduous tropical forest	tropical forest			
High-stature, evergreen tropical forest				
High-stature, semi-evergreen tropical forest				
Low-stature, evergreen tropical forest				
Low-stature, thorny, semi-evergreen tropical forest	Evergreen tropical	T1.1 Tropical/subtropical lowland rainforests		
Low-stature, semi-evergreen tropical forest	TOTEST			
Mid-stature, evergreen tropical forest				
Mid-stature, semi-evergreen tropical forest				
Crassicaulescent desert shrubland				
Tamaulipan thorny shrubland				
Xerophytic mezquital				
Chaparral	Woody vorophytic			
Coastal rosetophyllous shrubland	shrubland	T3.1 Seasonally dry tropical shrublands		
Sarcocaulescent shrubland	Siliubialiu			
Sarco-crassicaulescent shrubland				
Submontane shrubland				
Foggy sarco-crassicaulescent shrubland				
Microphyllous xerophytic scrubland				
Rosetophyllous xerophytic scrubland	Non-woody	T5 Deserts and semideserts		
Sandy desert vegetation	xerophytic scrubland	To Deserts and semideserts		
Halophytic xerophytic vegetation				
Gallery forest				
Gallery tropical forest	Woody bydrophylous	FT1 Palustrine wetlands		
Gallery vegetation	vegetation			
Petén vegetation	vegetation	MET1 Brackish tidal systems		
Mangrove forest				
Mezquite woodland				
Natural palm-dominated vegetation	Other woody	T1 Tropical-subtropical forests		
Induced palm-dominated vegetation	vegetation types			
Induced forest				



Cultivated pastureland		T7.1 Sown pastures and old fields		
Halophytic grassland				
Induced pastureland		T4 Savannahs and grasslands		
Natural grassland	Grassland			
Gypsophylous grassland	Grassianu			
Savannah				
Savannah-like vegetation				
High-mountain meadow		T6.5 Tropical alpine meadows and shrublands		
Popal vegetation	Non-woody			
Hydrophytic, halophytic vegetation	hydrophylous	FT1 Palustrine wetlands		
Tular	vegetation			
Sand dung vogstation	Other non-woody	TM2.1 Coastal shrublands and grasslands		
	vegetation			
Area devoid of vegetation	Other lands	N/A		
Areas without apparent vegetation	Other Idnus			

On the other hand, in order to integrate ecosystem information with socio-economic data, and better understand the flow of ecosystem services supplied by different ecosystems, it is also convenient to include non-ecological characteristics (*e.g.*, land use, land ownership, land management, etc.) in the delineation of ecosystem assets (UN, 2017; Bogaart *et al.*, 2019). The cross-classification of ecosystem and socio-economic information is key for assessing the relationship between ecosystem services, ecosystem assets and land management and ecosystem degradation. The link to socio-economic information can be better introduced during the process of delineating ecosystem assets by, for example, using information on land use or landownership, management regimes (*e.g.*, protected areas), etc.

Thus, a second set of criteria used for classifying Mexico's ecosystems aims to take into account differences in land management and use of natural resources stemming from the property regime (public, private or communal [including *ejido* or indigenous communities] property), as well as from restrictions imposed by the presence of (federal) natural protected areas. The classification of the country's ecosystems for the pilot tests of the NCAVES-Mexico project is, therefore, based on the intersection of three criteria:

- vegetation type or land use class, as *per* the CONAFOR's simplified version of INEGI's classification system
- land tenure (private *vs.* communal, including ejidos and indigenous communities), as *per* the registry kept by the National Agrarian Reform office, and
- presence of (federal) protected natural areas, as *per* the registry of the National Commission for Protected Areas.

Very recently, the SEEA-EEA revision process identified six design criteria that an ecosystem classification to be used in the SEEA-EEA framework should meet (Bogaart *et al.*, 2019), as follows:

- 1. The classification should be **based on ecological principles**.
- 2. The classification units should be **mappable**.
- 3. The classification should be **conceptually and geographically exhaustive**, able to describe and account for the entire surface of the study area.
- 4. Classification types should be **conceptually and geographically mutually exclusive**.



- 5. The classification should be **practicable**, having a hierarchical structure with a rather small number of universally understandable, high-order classes that allow for comparisons across countries to be made.
- 6. The classification should be **linkable** to other established classification systems.

The first four criteria should be met by any ecosystem classification that is to be used for ecosystem accounting at the national level; the other two criteria are rather necessary for standardizing country-level reports and between-countries comparisons (Bogaart *et al.*, 2019). As can be seen from the description above (pp. 5-6), the INEGI's classification system —as well as CONAFOR's simplified version of it— does neatly meet criterion 1 and its classes are defined (see Table 1 above and detailed descriptions in INEGI, 2014, 2017) in such a way that they are also conceptually exhaustive (criterion 3) and mutually exclusive (criterion 4). INEGI's classification system also neatly meets criterion 2: As discussed in section 5 below, using this classification system, INEGI regularly produces and updates 1:250,000 scale, wall-to-wall, country-wide charts portraying the geographic distribution of the different types of vegetation and land-use occurring in Mexico. Table 1 above also makes a first attempt to link (criterion 6) INEGI's and CONAFOR's classification systems with the corresponding high-order classes in IUCN's Global Ecosystem Typology (as described in Bogaart *et al.*, 2019).

Adopting INEGI's classification system as the basis for delineating Mexico's ecosystems for the pilot tests of the NCAVES-Mexico project is also in agreement with most of the guidelines derived from the SEEA-EEA revision process (Bogaart *et al.*, 2019). For example, INEGI's classification system does represent ecosystems defined with basis on —and characterized by— ecological properties and that are spatially delineable; it does encompass all the terrestrial ecosystems occurring in the country; the classes are mutually exclusive and exhaustive; and the classification has a well-defined hierarchical structure. Also, CONAFOR's simplified version of INEGI's classification provides a manageable ecosystem typology that does not have too many classes, thus reaching a balance between the number of different ecosystem types that are identified and the availability of information. In addition, while this classification is based on ecological principles, I am also explicitly including anthropogenic factors, such as land use and land ownership.

The approach followed for classifying Mexico's ecosystems in the NCAVES-Mexico project also coincides with one of the approaches suggested in the SEEA-EEA revision process for the construction of a national/regional ecosystem classification, namely, "...using as a starting point an existing national classification scheme, preferably, a classification that represents ecosystems or habitats, and that can be bridged to [a] reference...classification. This allows efficient use of available data, facilitates the integration of datasets and avoids producing partially overlapping datasets..." (Bogaart *et al.*, 2019).

3. Ecosystem extent

The second step in the SEEA-EEA framework is estimating the ecosystems' extent at different points in time. The ecosystem extent account provides the basis for the subsequent assessment of ecosystem condition and the flow of ecosystem services from each ecosystem type and asset. For most terrestrial ecosystems, extent is commonly measured in terms of surface area covered.

Using its own comprehensive classification system (see description in section 4 above), over the last 40 years INEGI has being producing and regularly updating 1:250,000 scale, country-wide, wall-to-wall charts portraying the geographic distribution and extent of the different types of vegetation and land-use occurring in Mexico. The charts also identify the areas without apparent vegetation and provide



general information on soil type and erosion, agricultural practices and crops, vegetation physiognomy, successional aspects and the use of plant communities, etc. The charts are produced by visual interpretation —by expert photo-interpreters— of satellite imagery (aerial photographs in the earliest versions), followed by on-the-ground verification. INEGI updates the 1: 250,000 scale Land use and Vegetation Charts every 4-5 years; six editions (known as *series*) have been published to date: Series 1 was produced based on the interpretation of aerial photographs recorded in the 1970's; Series 2 was based on satellite images (Landsat ETM 5) recorded in 1993; Series 3 was based on Landsat ETM7+ images recorded in 2002; Series 4 on Spot images recorded in 2007; for the Series 5, Landsat 5 images recorded in 2011 were used; and for the series 6, Landsat 8 images recorded in 2014.

Even though the various editions of INEGI's vegetation and land-use chart were made with different inputs and technologies, in all of them essentially the same vegetation classification scheme was used (with only a few minor differences between Series I and subsequent ones). Also important, the charts also provide information on the successional stage or conservation status (primary or relatively well preserved *vs.* second-growth or degraded) of vegetation, based on the presence of secondary species and the degree of soil erosion. Additionally, the charts include specific categories for anthropic (agriculture, livestock ranching, forestry, human settlements, etc.) or other land uses (*e.g.*, water bodies, areas devoid of vegetation, etc.). These features make INEGI's charts fully comparable with each other and allow examining, in a consistent manner, the extent and geographical distribution of Mexico's terrestrial ecosystems at different points in time (1970's, 1993, 2002, 2007, 2011 and 2014) with a spatial resolution (scale 1: 250 000) adequate for a reasonably detailed analysis of the whole country and of particular areas of it.

INEGI's vegetation and land-use charts have been recently criticized (see, for example, Gebhardt *et al.*, 2014) for their relatively coarse spatial resolution —their minimum mappable area is 25ha— and low update frequency (every 4-5 years), which are considered insufficient for a large country, with rapid vegetation dynamics, such as Mexico. Several efforts to construct more timely, wall-to-wall monitoring systems for the entire country have been carried out. To date, the most promising of these efforts is the Monitoring Activity Data for the Mexican REDD+ programme (MAD-MEX) system developed by CONABIO (Gebhardt *et al.*, 2014).

MAD-MEX is a fully automatized system for the classification of high-resolution satellite imagery (LANDSAT, Sentinel, and Rapid-Eye) based on field data from the National Forest Inventory, INEGI's vegetation and land-use charts and other ancillary data. The system produces land cover charts (scale 1:100,000, 1:50,000 and 1:20,000 respectively) annually, with a minimum mappable area of 1 ha or 0.1 ha respectively. The following products are currently available:

- a. Land cover 2015, scale 1:20,000 (5m resolution), based on Rapid-Eye imagery
- b. Land cover 2017, scale 1:50,000 (10m resolution), based on Sentinel imagery
- c. Land cover 2018, scale 1:50,000 (10m resolution), based on Sentinel imagery
- d. Land cover 2017, scale 1:100,000 (30m resolution), based on Landsat imagery
- e. Land cover 2018, scale 1:100,000 (30m resolution), based on Landsat imagery

The MAD-MEX 2015 1:20,000 chart (based on Rapid-Eye imagery) was independently reviewed and validated by experts and fine-tuned accordingly; this chart is known as the reference map. The other classification products (for 2017 and 2018) have not been similarly post-processed but were calibrated against the 2015 reference map; therefore, their reliability has not been actually assessed.



The MAD-MEX classification distinguishes, with decreasing degrees of accuracy, 8, 17, or 31 different land-cover classes, which partially correspond with some classes (or groups of classes) of INEGI's vegetation and land-use classification. While the temporal and spatial resolution of the MAD-MEX products make them very useful for various applications, it has been now recognized that, for ecosystem accounting purposes, land cover is insufficient for properly delineating ecosystem assets (Bogaart *et al.*, 2019).

For these reasons, and taking into account the general considerations described in section 3 of the main report, the extent of Mexico's ecosystems and its changes over time (as of 2002, 2007, 2011 and 2014) for the NCAVES-Mexico pilot projects were estimated based on INEGI's Vegetation and land use charts.

4. Ecosystem condition

4.1 Context

The third step in the SEEA-EEA framework consists in compiling information about the ecosystems' condition. Ecosystem condition is a fundamental component in the ecosystem accounting framework as it allows examining how the quality of ecosystems in the study area changes —due to natural or anthropogenic causes— over time (past trends and current state) and how those changes affect —in conjunction with the changes in ecosystem extent— their ability to supply ecosystem services. Such information can then be used to try to predict potential changes in the future. Together, the two accounts (*viz.*, ecosystem extent and ecosystem condition) help to identify ecosystem degradation or improvement, establish the link between ecosystem assets and the flow of ecosystem services, and can be used to support estimates of the monetary value of ecosystems and the services they supply (Keith *et al.*, 2019).

However, as of today there is no unique, widely accepted definition of ecosystem condition. The Millennium Ecosystem Assessment defined ecosystem condition in relation to its ability to provide ecosystem services (MEA, 2003). The OpenNESS Glossary defines ecosystem condition as the overall quality of an ecosystem, in terms of its main characteristics underpinning its capacity to generate ecosystem services (Potschin-Young *et al.*, 2016). A straightforward definition has been adopted by the working group on Mapping and Assessment of Ecosystems and their Services of the European Commission: Ecosystem condition refers to the physical, chemical and biological condition or quality of an ecosystem at a given point in time (Maes *et al.*, 2014, 2018). In the SEEA-EEA framework (UN, 2014, 2017) ecosystem condition is defined as the overall quality of an ecosystem asset in terms of its characteristics such as water, soil, carbon, vegetation and biodiversity. More recently, the SEEA-EEA revision has proposed a broader —albeit entirely pragmatic— definition: The condition of an ecosystem is interpreted as the ensemble of multiple relevant ecosystem characteristics, which are measured by sets of variables and indicators that, in turn, are used to compile the accounts (Keith *et al.*, 2019).

Not surprisingly, the measurement of ecosystem condition is still a matter of debate as well (Rendon *et al.*, 2019). The SEEA EEA framework has adopted a straightforward, general two-stage approach to assess the condition of ecosystem assets (UN, 2014, 2017; Keith *et al.*, 2019): In the first stage, a set of relevant ecosystem characteristics are selected and a set of measurements able to reflect changes in those characteristics are identified. In the second stage, the measurements are related to a corresponding reference condition to construct indicators. In order to apply, in practice, the general two-stage approach adopted by the SEEA EEA framework, it is necessary to identify:

a) the relevant ecosystem characteristics;



- b) the measurements to be used for each of those; and
- c) a reference condition for each measurement.

However, the SEEA-EEA framework (UN, 2014) and its technical recommendations (UN, 2017) provide only general recommendations on how to identify these elements in practice. First, with regard to the ecosystem characteristics to measure, these documents provide indicative examples of key ecosystem characteristics (water, soil, carbon, vegetation, and biodiversity); point out that the choice of characteristics will depend on the ecosystem types present in the study area and the ecosystem services being considered and that condition accounts should be compiled separately by ecosystem type; and recommend that ecosystem characteristics be selected on a sound scientific basis, to ensure that they reflect the ongoing functioning, resilience and integrity of the ecosystem.

Secondly, with regard to the measurements that should be used to describe changes in the ecosystem characteristics, the SEEA-EEA framework (UN, 2014) and its technical recommendations (UN, 2017) also provided only general recommendations: Measurements of ecosystem condition should be selected so that they reflect the general ecological state of an ecosystem, its capacity to supply ecosystem services and the relevant trends; variations in these measurements should be responsive to changes in the ecosystem's functioning and integrity. For each ecosystem characteristic, there will usually be more than one relevant indicator and, generally, different indicators will be necessary for different ecosystem types and for different ecosystem services. Ecosystem condition indicators may describe aspects such as the occurrence of species, soil characteristics, water quality, biodiversity, and ecological processes (*e.g.* net primary production) but the indicators selected should be relevant for policy and decision making, for instance because they reflect policy priorities, pressures on ecosystems, or the capacity of ecosystems to generate one or more services.

On the practical side, the technical recommendations (UN, 2017) also recognize that data availability (or the possibility to obtain additional data) is an important consideration in selecting condition indicators. Generally, condition indicators related to characteristics such as vegetation, water, soil, biomass, habitat and biodiversity for different ecosystem types, as well as indicators of relevant pressures and drivers of ecosystem change, will be appropriate. In some cases, composite indicators (combining the effect of different elements/aspects of ecosystem condition) may be useful. Five considerations to be used in selecting condition indicators are "...(i) the degree to which the indicator reflects the overall ecological condition of the ecosystem or key processes within it and is able to signal changes in this condition; (ii) the degree to which the indicator can be linked to measures of potential ecosystem services supply; (iii) how easy it is for policy makers and the general public to understand and correctly interpret the indicator; (iv) data availability and scientific validity of measurement approaches for the indicator; and (v) the possibility to generate new data cost effectively..." (UN, 2017).

Third, while the use of reference or benchmark conditions (and baselines) for indicators is well known and widely used, determining an appropriate reference condition for each indicator is not straightforward either (UN, 2014, 2017). At least two alternative approaches for determining the reference condition can be considered. One possibility is to assume the condition at the beginning of the accounting period as the reference condition and express changes in relation to that point in time. While this approach does suffice for accounting purposes, it entails the important shortcoming that it automatically assigns the same (best) condition to all the ecosystems at the beginning of the accounting period. An alternative approach is choosing as a reference condition that of a pristine, undisturbed ecosystem and then express changes in terms of the deviation from such condition. This approach allows different ecosystems to be compared fairly and its results can be readily interpreted. However, defining what a natural ecosystem looks like and what its conditions are is far from trivial.



In summary, although the general approach to assess the condition of ecosystems —in the context of the SEEA EEA framework— is well established and straightforward, a number of measurement issues and challenges in the compilation of ecosystem condition accounts still persists. As a result, early experiences with the SEEA-EEA framework have shown that different approaches to ecosystem condition accounting have been used in practice, leading to uncertainty about how these accounts should be compiled and used (Maes *et al.*, 2019). Selecting appropriate measures of ecosystem condition is a highly challenging task as this is an inherently multidimensional (and, as described above, not entirely welldefined) concept that, on the one hand, is expressed out in various different ecosystem characteristics and is, at the same time, the result of a number of external, ecological and non-ecological, factors affecting the ecosystem. Thus, assessing ecosystem condition involves examining several different characteristics related to the functioning and integrity of the ecosystem. In addition, different ecosystem condition measurements (e.g., water pH, soil nutrients, etc.) are relevant for different types of ecosystems and for different ecosystem services (e.g., soil fertility is directly relevant for food provision services, but less so for water supply, etc.). Other issues include the potential to aggregate across different characteristics to derive an overall index of condition, the level of spatial detail required, the selection of adequate reference conditions, and how to record changes in ecosystem condition over time (UN, 2017).

At the time of implementing the pilot tests of the NCAVES-Mexico project, more concrete guidance was found in the works by the working group on Mapping and Assessment of Ecosystems and their Services of the European Commission (Maes *et al.*, 2018). The MAES working group proposed a reduced set of ecosystem characteristics to consider as the basis for assessing ecosystem condition either per ecosystem type or across different ecosystems (see Table A.2 below). The MAES scheme distinguishes between environmental quality (which express the physical and chemical quality of ecosystems) and ecosystem attributes (biological quality of ecosystems); ecosystem attributes are further subdivided into structural and functional attributes. The ecosystem condition assessment framework proposed by the MAES working group also includes pressures on ecosystems as an important component: Given the strong causal relation between pressures and ecosystem condition, pressures can be used as approximate indicators of ecosystem condition in cases where direct indicators are not available.

Ecosystem condition					
Environmental quality (physical and chemical quality)					
Ecosystem attributes (biological quality)		Structural ecosystem attributes (general)			
	Structural ecosystem attributes	Structural ecosystem attributes based on species diversity and abundance			
		Structural soil attributes			
	Functional ecosystem	Functional ecosystem attributes (general)			
	attributes	Functional soil attributes			

Table A.2. Classification of ecosystem condition and pressure indicators, as proposed by the Mapping and Assessment of Ecosystem and their Services (MAES) working group. The original classification includes one additional class (*Structural ecosystem attributes monitored under the EU nature directives*) that only pertains to the European Union.



Pressures
Habitat conversion and degradation
Introduction of invasive alien species
Pollution and nutrient enrichment
Over-exploitation
Climate change
Other pressures

4.2 Selecting and developing ecosystem condition indicators for the NCAVES-Mexico project

Data availability was the main and first restriction faced when selecting and developing condition indicators for Mexico's ecosystems. An important finding of the *Mexico's country assessment on natural capital accounting and valuation of ecosystem services* was that, in general, there is insufficient data/information on ecosystem services that is solid, readily available, accessible, and has the necessary coverage and temporal and spatial resolution. For this reason, the implementation of the pilot studies of the NCAVES-Mexico project had to rely, mostly, on the statistical and geographical data/information that is regularly collected by various government agencies as part of their operation and which is relevant and — after suitable reformatting, adaptation, or reformulation—potentially useable for ecosystem accounting purposes.

Based on the results of the desk study on *Data availability and sources* included in chapter 3 of the *Mexico's country assessment on natural capital accounting and valuation of ecosystem services*, data sets potentially useful for developing condition indicators relevant/applicable to all or most of the terrestrial ecosystems occurring in the country and for the different classes included in the MAES scheme (see Table 2 above) were identified. Table A.3 below shows the variables and indicators that are included in (or can be derived from) the data sets thus identified. Each of these was then examined against the general criteria described in Section 3 of the main report (*i.e.*, that the necessary data should be available for the entire country, with adequate spatial and temporal resolution, for different points in time, etc.) to finally identify those most useful for the purpose. Results from this examination are described in detail in sections 4.4 – 4.9 below.

Table A.3. Variables, indicators and indices of ecosystem condition and pressure initially considered (shown in bold face) for Mexico's terrestrial ecosystems, for the four ecosystem services being assessed in the pilot studies of the NCAVES-Mexico project.

Ecosystem condition					
Environmental quality (physical and chemical quality)		Water-caused soil erosion			
Ecosystem Structural ecosystem attributes attributes		Structural ecosystem attributes (general)	Conservation status of vegetation Ecosystem Integrity Index		



(biological quality)		Structural ecosystem attributes based on species diversity and abundance Structural soil attributes	Biodiversity Ecological Integrity Index Soil organic carbon			
	Functional	Functional ecosystem attributes (general)				
ecosystem attribute		Functional soil attributes				
	Pressures					
Habitat conversion and degradation Human Footprint Index						
Introduction of invasive alien species						
Pollution and nutrient enrichment						
Over-exploitation						
Climate change						
Other pressures						

<u>4.3 Linking the ecosystem condition indicators of the NCAVES-Mexico project to the updated</u> guidelines from the SEEA-EEA revision process

Clearly, the very definition of ecosystem condition as well as the proper way to measure it in a form suitable for ecosystem accounting are challenges still found when compiling ecosystem accounts in practice. Further conceptual work, on-the-ground testing and experimentation, and the development of better measurement approaches/methods for this purpose are still necessary (UN, 2018). UNSD recently launched a process to revise the technical recommendations for developing ecosystem accounts within the SEEA EEA framework (UN, 2018). Among other research subjects, ecosystem characteristics and indicators of ecosystem condition were identified as revision issues requiring conceptual work and specific testing and experimentation. The revision of ecosystem condition accounts aimed, in particular, at (i) developing a generalized model or structure of characteristics and indicators of condition for different ecosystem types; (ii) providing a detailed and consistent set of criteria for selecting relevant ecosystem characteristics and condition indicators; (iii) determining whether nonecological characteristics, such as land use and management practices and pressures, should be included in condition; and (iv) providing a proposal for an indicator typology, *inter alia* (Keith *et al.*, 2019; Czúcz *et al.*, 2019).

Early results from the revision process have provided further clarification on the components of the condition assessment and additional detail to several key elements necessary for their practical implementation (Keith *et al.*, 2019; Czúcz *et al.*, 2019):

a) Definitions



- The <u>general approach to assess the condition of ecosystem assets</u> encompasses two stages. The first stage is to identify the most relevant ecosystem characteristics, and the second stage is to identify the appropriate quantitative measures to describe each of those characteristics
- * <u>Ecosystem characteristics</u> describe aspects such as vegetation, water, soil, biomass, habitat and biodiversity. Ecosystems have many quantifiable characteristics not all of which should be measured but the most relevant ones should be identified, and appropriate indicators be developed. Characteristics of ecosystem condition should not be restricted to those related to the provision of the final ecosystem services used by humans; rather, they should also include aspects related to the ecosystem's intrinsic values and the provision of intermediate ecosystem services.
- * <u>Variables, indicators, and indices</u> are concrete quantitative measures that can be used to describe the ecosystem characteristics identified.
 - <u>Ecosystem variables</u> are measurable quantities describing physical, chemical or biological characteristics of an ecosystem. Variables measure individual characteristics, but a single characteristic can be described by several variables. Changes in variables do not, by themselves, convey any value judgement as to being good or bad, beneficial or detrimental, etc.
 - <u>Reference level</u> is a value against which the current value of a variable is compared in order to derive an indicator. A reference level can reflect a natural state, a temporal baseline (the indicator value for a particular point in time), a desired value (the indicator value that a policy aims to achieve), a prescribed value (such as a legislated quality measure), or a threshold value (an indicator value above or below which there is evidence that ecosystem condition is sub-optimal).
 - <u>Indicators</u> are variables with a normative interpretation (*e.g.* a reference level) associated, with a view to informing policy and decisions. An indicator includes the values and their comparison with the appropriate reference level.
 - <u>Indices</u> are (thematically) aggregated indicators that, with a single figure, aim to describe broad aspects of the ecosystem. An <u>aggregate sub-index</u> combines several indicators describing a single characteristic of the ecosystem type. An <u>aggregate</u> <u>index</u> condenses all characteristics into a single indicator for a given ecosystem type, or one characteristic across ecosystem types.
- b) Selection criteria for ecosystem characteristics and indicators
 - As a tool to identify the relevant characteristics, variables and indicators to use in the assessment of ecosystem condition within the SEEA-EEA framework, a set of 12 selection criteria was formulated. Criteria applicable to individual variables and their derived indicators include: relevance, state orientation, framework conformity, spatial consistency, temporal consistency, feasibility, quantitativeness, reliability, normativity and simplicity. Criteria applicable to a whole set of indicators are parsimony and data gaps. All the criteria apply to the selection of variables and indicators, but only the first three ones (*viz.*, relevance, state orientation and framework conformity) are applicable to the selection of ecosystem characteristics should be made upon a more conceptual basis, whereas the selection of variables and indicators involves more pragmatic criteria.
- c) Typology of ecosystem condition



A general, hierarchical typology for ecosystem condition metrics is proposed, as an effort to provide a meaningful order among ecosystem condition indicators, facilitate communication, and provide a logical framework for comparisons across ecosystem types and the aggregation of different indicators into sub-indices or sub-indices into indices. The proposed typology includes:

<u>I Compositional (species-based) indicators</u> comprise a broad range of indicators describing the species composition of ecological communities. These include indicators based on the presence/abundance of a species or species group, or the diversity of specific species groups at a given location and time.

<u>II Structural (vegetation and biomass) indicators</u> describe the local amount of living and dead plant matter (vegetation, biomass) in an ecosystem, including all metrics of vegetation density and cover, either related to the whole ecosystem, or just specific compartments (aboveground, belowground, litter, etc.).

<u>III Functional (ecosystem processes) indicators</u> Encompass simple summary statistics (*e.g.* frequency, intensity) of relevant ecosystem functions.

<u>IV Indicators of abiotic characteristics (physical and chemical state)</u> Include various 'degradable environmental stocks' (*e.g.* soil organic carbon, tropospheric ozone, water table level, impervious surfaces) which directly change (deplete, accumulate) during a degradation process (*e.g.* erosion, pollution, desiccation, or soil sealing).

<u>V Landscape-level indicators</u> of ecosystem mosaics can describe the integrity of landscapes at broader spatial scales, including simple indicators of landscape diversity and connectivity/fragmentation.

d) Procedure

According to these new guidelines, in practice, condition accounts should be developed by first identifying —based on the selection criteria— relevant ecosystem characteristics within each class of the typology. Then, relevant variables to describe each of those characteristics are selected, also according to the selection criteria. Different combinations of these variables in association with reference levels are used to derive condition indicators. Some ecosystem condition indicators can then be combined with relevant ancillary data to derive indicators of the ecosystem's capacity to provide services

Based on the in-depth study of the data sets selected (listed in Table A.3 above) for developing condition indicators for Mexico's terrestrial ecosystems, an attempt was made to link them to the SEEA-EEA typology of ecosystem condition (see Table A.4 below). In addition, an initial appraisal of how well those variables and indicators meet the individual selection criteria of the SEEA-EEA framework was also attempted (see Table A.5 below). These findings are justified and discussed in the detailed descriptions given in sections 4.4 – 4.9 below.

Table A.4. Variables, indicators and indices initially considered for assessing the condition of Mexico's terrestrial ecosystems in relation to the new SEEA-EEA typology of ecosystem condition.

SEEA-EEA typology class	Variables	Indicators	Indices
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I. Compositional (species- based) indicators	Biodiversity		Ecological integrity index	
II. Structural (vegetation and biomass) indicators	Conservation status of vegetation		Ecosystem integrity index*	
III. Functional (ecosystem processes) indicators			Ecosystem integrity index*	
IV. Indicators of abiotic characteristics (physical and chemical state)	Soil organic carbon	Water- caused soil erosion		
V. Landscape-level indicators			Human footprint	
* The Ecosystem Integrity Index aggregates both, structural and functional variables. See full description in section 6.XX below				

Table A.5. Initial appraisal of whether or not or how well the variables, indicators and indices initially considered for assessing the condition of Mexico's terrestrial ecosystems meet the selection criteria of the SEEA-EEA framework.

	SEEA-EEA typology class						
	I. Compositional indicators		II. Vegetation and biomass		IV. Physical and chemical state		V. Landscape pattern
SEEA-EEA criterion	Biodiversity	Ecological integrity index	Conservation status of vegetation	Ecosystem integrity index	Soil organic carbon	Water- caused soil erosion	Human footprint
Relevance	Yes	?	Yes	Yes	Yes	Yes	Yes
State orientation	Yes	Yes	Yes	Yes	Yes	Yes	No
Framework conformity	Yes	Yes	Partly	Yes	Yes	Yes	Partly
Spatial consistency	Yes	Yes	Yes	Yes	Partly	Yes	Yes
Temporal consistency	No	No	Yes	No	No	No	Yes
Feasibility	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Quantitativeness	Yes (ratio)	Yes (ordinal)	Yes (ratio)	Yes (ordinal)	Yes (ratio)	Yes (ordinal)	Yes (ordinal)
Reliability	High	Low	High	Moderate	High	Moderate	High
Normativity	No	Yes	Partly	Yes	No	Yes	Yes
Simplicity	Yes	No	Yes	No	Yes	Yes	No

<u>4.4 Conservation status of vegetation</u>.- INEGI's Land Use and Vegetation 1:250,000 scale charts distinguish the successional stage or conservation status (primary or relatively well preserved *vs*.



second-growth or degraded) of vegetation, recognizing three different phases (or successional stages) of second-growth vegetation:

- Tree-dominated, second-growth vegetation,
- Shrub-dominated, second-growth vegetation, and
- Predominantly herbaceous second-growth vegetation

INEGI's Vegetation and land-use charts are freely available for six different points in time (*ca.* 1976, 1993, 2002, 2007, 2011 and 2014). From the information contained in these charts, the extent of the country's terrestrial ecosystems with different conservation status and its changes over time can be readily examined both in tabular (see example in Fig.A.1) and in a spatially explicit manner.



Fig.A.1 Changes over time in the conservation status (primary and well-preserved vs. degraded or second-growth) of vegetation in Mexico's terrestrial ecosystems, as estimated from INEGI's Vegetation and land-use charts for 2002, 2007, 2011 and 2014.

The conservation status of vegetation actually comprises two interlinked variables (extent of well-preserved and extent of degraded vegetation, in ha) that describe relevant structural characteristics of the natural terrestrial ecosystems occurring in Mexico (not applicable to agroecosystems or urban ecosystems). These variables clearly meet all the general criteria described in Section 3 of the main report, as well as most of the SEEA-EEA selection criteria for individual indicators (see Table A.5 above), except for normativity and framework conformity. With regard to the normativity criterion, the difficulty lies in setting an appropriate reference level. A pragmatic solution could be setting the extent of each condition (well-preserved vs. degraded) at the beginning of the accounting period as the reference level (separately for each ecosystem type). From an intrinsic point of view, however, a better approach would be setting the extent of primary vegetation (for each ecosystem type) in the absence of (or before) any anthropic impacts as the reference level —which is not easily determined. With regard to the framework conformity criterion, an important issue is that these two variables are inextricably linked to ecosystem extent, which might make it difficult to interpret changes in the levels of these variables, if not examined in relation to the concomitant changes in ecosystem extent.



<u>4.5 Soil erosion</u>.- Based on visual interpretation of satellite SPOT images, followed by on-the-ground verification, INEGI produced a hydric soil erosion, 1:250,000 scale, chart. The chart nicely depicts the extent, the degree (in relative terms), and geographical distribution of water-caused soil erosion in the country as of the period 2010-2015 (see Fig. A.2). More importantly, by combining the information provided by this chart with that contained in INEGI's vegetation and land-use chart for the relevant (or closest) period of time (*e.g.*, 2011), the extent and severity (in relative terms) of soil erosion affecting Mexico's terrestrial ecosystems can be readily examined (see Fig. A.3 below).



Fig.A.2 INEGI's water-caused soil erosion, 1:250,000 scale chart for 2010-2015. The chart depicts the relative degree (stable surface, light, moderate, strong or severe) of erosion affecting the different soil units in the country.





Fig.A.3 Relative level of soil erosion in Mexico's terrestrial ecosystems, as estimated from INEGI's Water-caused soil erosion chart (2010-2015) and INEGI's Vegetation and land-use chart for 2011.

The information contained in INEGI's hydric soil erosion chart provides a straightforward indicator of a relevant abiotic characteristic of Mexico's terrestrial ecosystems (including agroecosystems) and includes an in-built reference level (stable surface). The data available for this indicator meet most of the general criteria described in Section 3 above (except for availability of a consistent time-series), as well as most of the SEEA-EEA selection criteria for individual indicators (see Table A.5 above), except for temporal consistency and, partly, for reliability. With regard to the reliability criterion, the difficulty lies in the fact that erosion severity is not actually measured but only visually assessed (using an ordinal scale) by experts, which introduces some level of subjectivity in the indicator values. At present, however, the most significant shortcoming of this indicator is that INEGI has so far only produced one edition (for the period 2010-2015) of this chart and producing a new, updated edition is not yet contemplated. Nevertheless, whenever INEGI updates this chart, the new data should be readily incorporated into Mexico's ecosystem condition accounts.

<u>4.6 Soil organic carbon</u>.- Information on the content of organic C in the soil's top horizon can be readily obtained from the vast archive of soil data that INEGI (as part of its national soil mapping programme) and CONAFOR (as part of its National Forest and Soils Inventory programme) have been compiling. As of today, data available come from a total of 21 806 soil profiles sampled throughout the country over a long period of time (INEGI data: since 1968 to date; CONAFOR data: 2009-2014) and using different methods (see example in Fig.A.4 below).





Fig.A.45 Geographical distribution of part of the soil profile data set collected by INEGI as part of their national soil mapping programme. The figure shows the distribution of the 4,428 soil profiles used for the construction of INEGI's soil chart series II, scale 1:250,000.

Soil organic carbon is a key abiotic characteristic of terrestrial ecosystems (including agroecosystems). From this variable, relevant indicators of the condition of Mexico's terrestrial ecosystems could, in principle, be derived. Unfortunately, as of today, the data available for this variable do not meet key selection criteria (see Table A.5 above), including temporal and spatial consistency; they do not meet the normativity criterion either. With regard to the spatial consistency criterion, although all the data points are fully georeferenced, in order for them to provide a comprehensive, spatially-explicit description, they would have to be first made comparable and, then, spatially interpolated in order to produce a map of the variable (or of any derived indicator). Several efforts have been made to consolidate, make comparable and map this vast data set either directly (*e.g.*, Cruz-Gaistardo & Paz-Pellat, 2013; Paz Pellat *et al.*, 2016) or through the use of advanced geostatistical modelling techniques (Guevara *et al.*, accepted). However, to date, no agreed-upon, sufficiently accurate method has been produced yet. An initiative recently launched by UNSD and INEGI to implement in Mexico the S-World approach (Stoorvogel *et al.*, 2017) for modelling soil properties in a spatially-explicit manner using local (tier 2) data might be able to help to fill this important data gap.

At present, however, the most significant shortcoming of the data available for this variable is that they are, in reality, a mixture of recent and legacy data collected over a long period of time, so that they cannot be referred to a specific point in time (temporal consistency criterion). Periodically updating the data in the future (availability of a time series in section 3) is a closely related issue for which no simple solution exists either, as this necessarily demands specific field-sampling efforts for monitoring the soil variables across the country.

Mexico has not yet reported on indicator 15.3.1 Proportion of land that is degraded over total land area of Sustainable Development Goal 15. Life on Earth (see Gobierno de la República, 2018). This indicator encompasses three sub-indicators, namely, Land cover change, Net primary productivity, and



Soil Organic Carbon Stocks. Information on land cover change are readily available in Mexico but, as shown here, not so for the sub-indicator on soil organic carbon.

<u>4.7 Biodiversity</u>.- CONABIO's National System of Biodiversity Information (SNIB, for its Spanish acronym) contains a vast archive of fully-georeferenced collection records of plant and animal specimens in Mexico. The SNIB contains almost one million collection records (see Fig. A.5 below) that document in detail the presence of 14,424 different species of higher plants across the country, distinguishing between endemics and species at risk.



Fig.A.5 Geographical distribution of 887,242 unique collection records of higher plants in Mexico, as available in the National System of Information on Biodiversity. Notice the biased distribution of sampling points, strongly clustered along highways.

From this vast data set, relevant variables and indicators (*e.g.*, species richness and number of species at risk per ecosystem type) of the condition of Mexico's terrestrial ecosystems could, in principle, be derived. Unfortunately, the SNIB data actually come from voucher specimens stored in various Mexican and international herbaria, the specimens having been collected throughout the country over a very long period of time (since the late XVI century to present). Thus, as is also the case with soil organic carbon, this legacy data cannot be referred to a particular point in time (temporal consistency criterion). Periodically updating the data in the future (availability of a time series in section 3) is another closely related issue that cannot be easily solved.

Another shortcoming in the data available for these variables is that the samples distribution is strongly biased/limited by accessibility conditions (spatial consistency criterion). Although all the sampling points are fully georeferenced, in order for them to provide a comprehensive, spatially-explicit description, they would have to be first spatially interpolated in order to produce a consistent map of the variable (or of any derived indicator). Some efforts have been made to use these data sets to map the potential distribution of species by means of geostatistical models (*e.g.*, Ceballos *et al.*, 2006) but these have been limited to only some species.



In summary, the legacy collection records data currently available from CONABIO's SNIB are not suitable to construct a species-based biodiversity indicators. CONABIO (together with INECOL and other institutions) recently (as of 2014) launched an effort to set up a National System for Biodiversity Monitoring, which is now collecting observational data (by means of camera and sound-recording traps) on species occurrences at a number of sampling sites across the country (mostly located in protected areas). The data collected by the SNMB will be more suitable for this purpose.

<u>4.8 Aggregated indices of ecosystem characteristics</u>.- Recent studies have attempted to integrate several variables into composite indicators or indices that aim to provide, in a single figure, a measure of the overall condition of ecosystems. While integrating several related variables into a single index allows for easier communication and interpretation, the construction and use of indices also entails two important difficulties: Indices combine or lump together several different variables which may be independent of, or partially correlated with, each other. Thus, changes in the index value have no easily discernible meaning or interpretation without revisiting its component variables. Secondly, in most cases there is no natural or sound way to assign relative weights to the various variables contributing to the index value. Any approach undertaken to form the index involves making often untested assumptions about the relative importance of each variable and the correlations among them.

Two such indices have been recently implemented for Mexico.

<u>4.8.a The Ecosystem Integrity Index (INECOL-CONABIO)</u>.- In 2014, the National Commission for the Knowledge and Use of Biodiversity (CONABIO), together with the National Forestry Commission (CONAFOR) and the National Commission of Natural Protected Areas (CONANP) launched a coordinated effort to design and implement a National System for Biodiversity Monitoring (SNMB) in Mexico. The SNMB aims to monitor the state of the country's biological diversity and its changes over time, throughout the national territory. One of the key components of the SNMB is the Ecosystem Integrity Index (EII), a synthetic index to monitor the condition (or "health status") of the country's ecosystems. The EII was jointly designed by CONABIO and INECOL specialists (Garcia-Alaniz *et al.*, 2017), as part of the activities of the *Role of Biodiversity in Climate Change Mitigation* (ROBIN) project funded by the European Union.

The EII seeks to reflect the structural integrity of ecosystems in a single figure. In this approach, ecosystem integrity is evaluated in terms of how different an actual ecosystem is from some original or desired condition. The underlying idea is that by measuring compositional, structural and functional attributes of ecosystems, Ecosystem Integrity can be inferred and assessed— in relative terms with respect to the original, unimpaired condition.

The EII is constructed from two types of information (physical-chemical conditions and structural and functional attributes) that, together, determine the condition of the ecosystem. Data from different sources are used to quantify or estimate these components. Ecosystem structure is evaluated based on field measurements of forest structure variables (*e.g.*, average tree height, average DBH, average canopy diameter, proportion of dead trees standing, tree density, etc.) that CONAFOR periodically collects throughout the country as part of its National Forest Inventory. These point data (and additional variables derived from them, such as the standard or the absolute deviation of tree height, etc.) are then spatially interpolated by building statistical models using satellite, topographic and climatic data as predictor variables.



Functional features of ecosystems (such as annual gross primary productivity, annual net photosynthesis, etc.) are evaluated using estimators derived from satellite imagery (MODIS). Holdridge life zones and site elevation are used to infer the values expected under relatively unimpaired conditions, given the physical-chemical conditions of the site. A third group of variables reflects factors that can affect the condition of ecosystems, such as the presence of human settlements, fields, pastures, etc. All the variables are rated using a single 0 to 1 score, relative to the unimpaired conditions. All the structural and functional variables are then integrated into a complex statistical model (Bayesian network) to produce the EII (see conceptual scheme in Fig. A.6 below), which values can be represented in the form of maps (see example in Fig.A.7 below).



Fig.A.6 Conceptual model of the Ecosystem Integrity Index. The physicochemical setting present in a given area (bottom) establishes the context for the composition, structural and functional attributes (middle) of the ecosystem. These are associated with a certain ecosystem integrity condition (top) (taken from García-Alaniz *et al.* 2017).

The SNMB began monitoring the condition of the country's ecosystems using the EII in 2015. CONABIO and INECOL have produced yearly country-wide charts of the EII with a 1km spatial resolution for 2004 through 2014. More recently, a 250m spatial resolution data set for the year 2014 was also released (see Fig. A.7 below). All these data sets are freely and openly available. CONABIO and INECOL are currently working on incorporating (species-based) compositional data into the IIE, using observational data recorded by camera- and sound-recording- traps. In addition, plans to produce 250m resolution data sets for other dates besides 2014 are in course.





Fig.A.7 Ecosystem Integrity Index for 2014. The EIII aims to reflect the structural integrity of ecosystems in a single figure. Values closer to 1 denote ecosystems relatively whose structure, functioning and composition is relatively unimpaired.

The EII is an aggregated index of relevant structural and functional characteristics of Mexico's terrestrial ecosystems (including agroecosystems) and includes an in-built reference level (an unimpaired original condition). The EII meets most of the general criteria described in Section 3 above (except for availability of a consistent time-series), as well as most of the SEEA-EEA selection criteria for individual indicators (see Table A.5 above), except for temporal consistency, simplicity and reliability.

With regard to the simplicity criterion, although the EII constitutes an attractive, useful communication tool, that can be easily communicated to non-specialized audiences, in reality it lumps together (by means of a complex statistical model) several different variables. Thus, the EII values (and its changes) are not easily interpretable; it is necessary to trace back the EII values/changes to the component variables in order to relate and interpret them in relation to ecosystem services.

With regard to the reliability criterion, the problem lies in the fact that even though the EII uses a substantial amount of primary data (field collected by the National Forest Inventory), these have to be first spatially interpolated by means of geostatistical models. More importantly, the EII also incorporates a number of estimators of functional features of ecosystems (such as annual gross primary productivity, annual net photosynthesis, etc.) which are not actually measured but derived from satellite imagery (MODIS). These factors add to the uncertainty of the EII values.

The lack of a consistent time series for the EII seems to be only a temporary limitation for the use of this indicator, as CONABIO e INECOL are planning to produce 250m resolution data sets for other dates besides 2014 in the near future.



From a more conceptual point of view, one other shortcoming of the EII that might be worth considering is that, being based on a number of forest structure variables, the index values are naturally biased towards forested ecosystems and tend to undervalue other key ecosystems such as natural grasslands, scrublands, etc. (personal observation, data available upon request). Further, closer examination of the distribution of the EII values across different ecosystems is necessary and perhaps its use as an index of ecosystem condition be limited to only forested ecosystems.

<u>4.8.b Ecological Integrity Index</u>.- The Ecological Integrity Index (EII') proposed by Mora (2017) aims to characterize "...the potential of natural landscapes to support ecological integrity in maintaining biotic and abiotic apex predators' interactions...". Conceptually, the EII' is based on an ecological hierarchical network as a framework to describe changes occurring at several levels in an ecological hierarchy (see Fig. A.8 below). Accordingly, the methodological approach includes: "(a) the construction of spatially explicit manifest indicators for ecological integrity as the foundation for the evaluation system; (b) the application of structural equation models for deriving a set of latent indicators that build the notion of ecological integrity at two levels (1st and 2nd order latent indicators); and finally, (c) a general indicator that summarizes the integrity in the ecological condition" (Mora, 2017).



Fig.A.8 The Ecological Integrity hierarchy framework for evaluating the condition of natural ecosystems based on landscape characteristics that sustain predator-prey interactions (taken from Mora, 2017).

The EII' implemented by Mora (2017) was constructed based on previously constructed statistical models of the potential distribution of 232 mammal species and 7 top predators (Ceballos *et al.*, 2006), along with data from INEGI's vegetation and land-use chart series IV. By overlapping and doing simple operations with the species distribution models, eight so-called "manifest ecological integrity indicators" (functional diversity, predator/prey diversity, habitat specialization, habitat selection, remnant habitat, habitat connectivity, trophic connectivity, and network resistance) were calculated. The manifest indicators were then used to build spatially-explicit Structural Equation Models for seven so-called "latent (or abstract) indicators of ecological integrity": Self-organization, Stability, Naturalness, Biodiversity, Mobile links, Spatial intactness, and Landscape heterogeneity. The latent indicators are finally aggregated into an Ecosystem Integrity Index (Mora, 2017). A country-wide chart of the EII' with a 1km spatial resolution for an un-specified date (see Fig.A.9 below) is available upon request.

The 1km spatial resolution of the EII' data set is relatively coarse for the purposes of the NCAVES-Mx project and no consistent time-series data are available (see general



considerations in Section 3 above). The EII' meets most of the SEEA-EEA selection criteria for individual indicators (see Table A.5 above), except for temporal consistency, reliability and simplicity.

With regard to the simplicity criterion, the EII' lumps together a number of different variables and synthetic indicators into a single figure, which makes its values difficult to interpret and relate to field conditions. Thus, the EII' values (and its changes) are not easily interpretable and, given the complexity of its construction process, it would be very difficult to trace back the EII' values to the underlying component variables.



Fig.A.9 Ecological Integrity Index. The EII' aims to evaluate the condition of natural ecosystems based on landscape characteristics that sustain predator-prey interactions.

With regard to the reliability criterion, the problem lies in the fact that, except for the information derived from INEGI's Vegetation and Land-use chart, all the other components of the EII' are model estimates/projections and no actual measurements. The EII' is constructed by overlapping ("stacking") a number of potential distribution models for individual species and, from this, predator-prey interactions are claimed to be inferred. While the use of stacked species distribution models has been increasing lately as a modelling tool to examine biodiversity patterns and other issues, there have been repeated calls to first —as with any predictive empirical model— validate the models through an evaluation of their predictive performance prior to further use (*e.g.* Fielding and Bell, 1997; Gastón & García-Viñas, 2013). While the species distribution models used in the construction of the EII' were built from actual field collection/sighting records, they do not seem to have been properly validated (see Ceballos *et al.*, 2006) nor has the EII' model itself. All these factors add to the uncertainty of the EII' values.

Finally, with regard to the spatial consistency criterion, the same fact that the EII' is constructed essentially from model projections —rather than actual measurements— implies that its values cannot be referred to a specific point in time and cannot be updated.

4.9 Indicators of socio-economic pressures and drivers of change.- As can be seen from the results above,



the major limitation faced when compiling ecosystem condition accounts for Mexico's ecosystems is the scarcity of relevant data with the required qualities (see section 3 in the main report and Table A.5 above). On this regard, the ecosystem condition assessment framework proposed by the MAES working group considers that, given the strong causal relation existent between pressures and ecosystem condition, in cases where direct indicators of ecosystem condition are not available, pressure indicators can be used as approximate indicators of ecosystem condition. Thus, the possibility of using variables and indicators of the socio-economic pressures and drivers of change affecting Mexico's ecosystems was then explored.

Abundant data are available on a range of human activities affecting Mexican ecosystems, including infrastructure works, agriculture and livestock ranching, urban development, tourism development, mining, etc. However, some of these activities can often physically coincide, fully or partially, in some locations or times, and it is their aggregated impact what really affects the ecosystems functionality. This aggregated impact can hardly be captured if the various activities are examined separately, in isolation from each other. Thus, an integrated approach that aggregates the combined impacts of the various activities occurring in the same place into a composite index might be preferable.

Since the launching of the Ecological Footprint index in the mid 1990's (Wackernagel & Rees, 1996), there have been several initiatives to compile indices denoting the physical impact that the range of human activities have on the structure/integrity of ecosystems. Two of the best known, global-level attempts to construct spatially-explicit indices of this type were carried out by Sanderson *et al.* (2002) and, more recently, by Venter *et al.* (2016). The first implementation of a Human Footprint for Mexico was by González-Abraham *et al.* (2015), which depicts conditions as of 2011. More recently, SEMARNAT made a more comprehensive implementation of the Human Footprint for Mexico, also for the year 2011, which was included in the 2015 edition of the Mexico's State of the Environment Report. In response to an *ad hoc* request by the NCAVES-Mexico project, SEMARNAT decided to produce an updated version of the index with data for 2014-2015.

The Human Footprint index (HF) is based on previous work by Bonham-Carter (1994) and González-Abraham *et al.* (2015). The indicator denotes, in relative terms, the extent to which natural environments have been modified or transformed by human activities. It is computed by estimating/assessing both, the Extent and Intensity of the transformation caused by various human activities —for which spatially explicit information is available— including:

- Cities and towns (< 500 inhabitants, 500 2500 inhabitants)
- Agriculture and aquaculture; forest plantations; cultivated pastureland
- Roads (highway, dirt-road, carpeted road, gravel road), railways, electricity transmission lines
- o Industry
- Wastewater treatment facilities
- o Artificial salt flats
- Archaeological sites
- o Solid waste final disposal sites (dump sites, sanitary landfills)
- Mines (primary, secondary, tertiary zones)

The <u>extent</u> of the transformation induced by each activity is expressed in terms of the area covered by the activity; for linear features (*e.g.*, transmission lines) 100m or 250m wide buffers were considered, for point features (*e.g.* archaeological sites) a 500x500m extent was imputed to each of them. The <u>intensity</u> of each activity was expressed in relative terms using a three-point ordinal scale (5, low; 7, medium; and 10, high intensity), depending on the perceived severity and duration of the impact (*e.g.*,


urban areas -> high intensity; agriculture -> medium intensity, forestry -> low intensity, etc.).

The HF index is computed based on actual, spatially explicit data (spatial scales ranging from 1:50,000 to 1:250,000) that are compiled and updated regularly or at irregular intervals by relevant government agencies including INEGI, SEMARNAT, Secretaría de Comunicaciones y Transportes, CONAGUA, Instituto Nacional de Antropología e Historia, etc. SEMARNAT has so far produced two country-wide HF maps for 2011 and 2014-2015, with a 500m spatial resolution. Based on this information, the geographical distribution, extent and severity of the impact of human activities on the country's ecosystems (and their changes over time) can be readily examined (see example in Figs. A.10 and A.11 below).



Fig.A:10 Geographic distribution of Human Footprint values in Mexico as of 2011.





Fig.A.11 Changes over time (2011 to 2014/2015) in the impact of human activities on the main ecosystems in Mexico.

As can be seen, the HF index meets all the general requirements stated in section 3 above. It can be updated whenever new information on infrastructure and human activities occurring in the country becomes available. In addition, when (spatially-explicit) information on other not yet considered activities becomes available, the index can readily incorporate it. On the other hand, the HF lumps together a number of different variables denoting the influence of various human activities that impact or modify natural landscapes. Its main shortcoming is that its values (and its changes) cannot be readily interpreted (or translated into actionable terms), as a given change might be the result of changes in one or several of its component variables.

On the other hand, however, the HF index clearly violates two of the main SEEA-EEA selection criteria, namely, state orientation and framework conformity. In fact, contrary to the view adopted by the MAES working group and a rather extended practice (e.g., Rendon et al., 2019; Maes et al., 2019), the SEEA-EEA criteria specifically recommends avoiding the use of pressure and other non-ecological variables (e.g., protection status, land management, etc.) in ecosystem condition accounts. Instead, the recommendation is to identify the underlying 'degradable stocks' affected by pressures and use these as condition indicator. While the rationale for this strict requirement holds, it overlooks some important facts both, practical and conceptual. On the one hand, the actual impact of environmental pressures on ecosystems often cannot be easily captured by one single variable, particularly when several environmental pressures act simultaneously on the same ecosystem or place. In such cases, the pressures on the ecosystem may be known, but the combined effects on the ecosystems' functioning and condition are often not well understood. As the total impact cannot be directly measured, it would, at least can be accounted for by considering the various pressures impinging on the same site. On the practical side, it is true that environmental pressures are often used in cases where data are not available but this is very often the case particularly in not so-data-rich countries. Excluding the use of environmental pressure indicators altogether might restrict severely the scope of the analyses. In addition, restricting the analysis of condition to state indicators might make its findings less relevant in practice, as policy responses cannot operate directly on state variables but they do so through regulating, eliminating or ameliorating pressures.

5. Conclusions and lessons learned



<u>Data availability</u>. Clearly, the first and major issue faced when compiling ecosystem condition accounts for Mexico's ecosystems was the availability of sound, reliable data that meet all the necessary requirements (general considerations in Section 3; SEEA-EEA selection criteria in Table 5). For example, for the pilot projects of the NCAVES-Mexico project, only seven variables/indicators/indices were initially identified based on data availability. Upon closer examination, fully suitable data (*i.e.*, meeting the various requirements/criteria set in section 3 above) were only available for only two of those, namely, the conservation status of vegetation, the Human Footprint Index. Data currently unavailable for the Ecosystem Integrity Index and soil erosion might be produced or made available in the near future. However, suitable data on key variables such as soil organic carbon content and biodiversity will be more difficult to obtain.

Vast amounts of relevant data are available in Mexico, obtained through soil surveys, the national forest inventory, the national system of information on biodiversity, and other national programmes, but having been compiled for purposes other than ecosystem accounting, those data sets do not readily meet the requirements necessary to be used for this purpose. More efforts are needed to collect/compile country-wide, spatially explicit, moderate resolution, multi-date data on biophysical variables/indicators of ecosystem condition, using existent and traditional data sources (*e.g.*, field monitoring), data from other sources (*e.g.*, remote sensing), and suitable modelling tools (*e.g.* S-World).

For these reasons, the push to avoid the use of pressure indicators to assess ecosystem condition might be overly restrictive in practice in the rather common cases where suitable data on direct indicators are not available.

<u>Variables, indicators and indices</u>. Being measures of biophysical variables, variables/indicators of ecosystem characteristics such as soil organic carbon, soil erosion, biodiversity, etc. might not always be easily interpreted by non-specialists and need specialized knowledge for interpretation. Nevertheless, their values and changes can be more readily related to particular ecosystem services and to the ecosystem's capacity to supply them. On the other hand, indices (composite indicators) often constitute attractive, useful communication tools, as they can be easily communicated and interpreted by non-specialized audiences. However, as they lump together data from several different variables, the meaning of their values (and their changes) are not immediately clear. It is often necessary to trace back the values/changes to the component variables in order to relate and interpret them in relation to ecosystem services and translate those findings into actionable information. For example, although the two indices examined aimed to evaluate the integrity of ecosystems, they use different aspects of variables, namely, forest structure and function in the case of the EII vs. predator-prey interactions in the case of the EII'. Thus, these two equally-named, equally-scaled indices in reality reflect different aspects of ecosystems. Indices lump together several different variables and there is no clear, univocal, objective way to assign weights to each component.

6. References

Aitor Gastón, Juan I. García-Vinas. 2013. Evaluating the predictive performance of stacked species distribution models applied to plant species selection in ecological restoration. Ecological Modelling 263 (2013) 103–108

Ceballos, G., S. Blanco, C. González, E. Martínez, (2006). Proyecto: DS006, Modelado de la distribución de las especies de mamíferos de México para un análisis GAP. Con un tamaño de píxel: 0.01 grados decimales. El proyecto fue financiado por la Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. México.



Czúcz, B., Keith, H., Jackson, B., Maes, J., Driver, A., Nicholson, E., Bland, L. (2019) Discussion paper 2.3: Proposed typology of condition variables for ecosystem accounting and criteria for selection of condition variables. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 13 March 2019. 23 pp.

Fielding, A.H., Bell, J.F., 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24, 38–49

Flores Mata, J., L. Jiménez López, X. Madrigal Sánchez, F. Moncayo Ruiz y T.F. Takaki. 1971. Mapa y descripción de los tipos de vegetación de la República Mexicana. Dirección de Agrología, Secretaría de Recursos Hidráulicos, México

González Medrano, F. 2003. Las comunidades vegetales de México. Propuesta para la unificación de la clasificación y nomenclatura de la vegetación de México. INE-SEMARNAT. México.

González-Quintero, L. 1974. Los tipos de vegetación de México, en: El escenario geográfico. INAH, México, pp. 109-218.

INEGI 2014. Diccionario de datos de uso del suelo y vegetación. Escala 1:250,000 (versión 3). Instituto Nacional de Estadística y Geografía – México.

INEGI. 2017. Guía para la interpretación de cartografía: Uso del suelo y vegetación, escala 1:250,000, serie VI. Instituto Nacional de Estadística y Geografía. México

Leopold, A.S. 1950. Vegetation zones of Mexico. Ecology 31:507-518

Maes J, Teller A, Erhard M, et al. (2014). Mapping and Assessment of Ecosystems and their Services. Indicators for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. Publications office of the European Union, Luxembourg.

Maes J, Teller A, Erhard M, Grizzetti B, Barredo JI, Paracchini ML, Condé S, Somma F, Orgiazzi A, Jones A, Zulian A, Vallecilo S, Petersen JE, Marquardt D, Kovacevic V, Abdul Malak D, Marin AI, Czúcz B, Mauri A, Loffler P, Bastrup-Birk A, Biala K, Christiansen T, Werner B (2018) Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem condition. Publications office of the European Union, Luxembourg.

Maes, J., Driver, A., Czúcz, B., Keith, H., Jackson, B., Bland, L., Nicholson, E., Dasoo, M. (2019) Discussion paper 2.2: Review of ecosystem condition accounting case studies: Lessons learned and options for developing condition accounts. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 13 March 2019. 25 pp.

Millennium Ecosystem Assessment (MEA) 2003. Ecosystems and human well-being: A framework for assessment. World Resources Institute. Washington, D.C. USA

Miranda, F., y E. Hernández Xolocotzi. 1963. Los tipos de vegetación de México y su clasificación. Boletín de la Sociedad Botánica de México 28:29-179.



Potschin-Young M, Haines-Young R, Heink U, Jax K 2016. OpenNESS Glossary. <u>http://www.openness-project.eu/glossary</u>

Paula Rendon, Markus Erhard, Joachim Maes & Benjamin Burkhard (2019) Analysis of trends in mapping and assessment of ecosystem condition in Europe, Ecosystems and People, 15:1, 156-172.

Rzedowski, J. 1978. Vegetación de México. Limusa, México.

Sánchez-Colón, S., A. Flores-Martínez, I.A. Cruz-Leyva & A. Velázquez. 2009. Estado y transformación de los ecosistemas terrestres por causas humanas. In: Capital Natural de México Vol. II: Estado de conservación y tendencias de cambio. CONABIO. México, D.F. pp. 75-129.

United Nations 2014. System of Environmental-Economic Accounting 2012—Experimental Ecosystem Accounting. United Nations publication, Series F, No.112. United Nations

United Nations 2017. Technical recommendations in support of the System of Environmental-Economic Accounting 2012. Experimental Ecosystem Accounting. White cover publication, preedited text subject to official editing

United Nations (2018) SEEA Experimental Ecosystem Accounting Revision 2020. Revision Issues Note.

Velázquez, A., Medina García, C., Durán Medina, E., Amador, A., Gopar Merino, L. F.. 2016. Standardized Hierarchical Vegetation Classification. Mexican and Global Patterns. Springer International Publishing. Switzerland. 143p.

Gobierno de la República. 2018. INFORME NACIONAL VOLUNTARIO PARA EL FORO POLÍTICO DE ALTO NIVEL SOBRE DESARROLLO SOSTENIBLE Bases y fundamentos en México para una visión del desarrollo sostenible a largo plazo. Avance en el cumplimiento de la Agenda 2030 y los Objetivos de Desarrollo Sostenible. México.

