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Implementing Ecosystem Natural Capital Accounting Methodology to the Rhone watershed: the proof-of-concept

Jazmin Adriana Argüello Velazquez

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Jazmin Adriana Argüello Velazquez. Implementing Ecosystem Natural Capital Accounting Methodology to the Rhone watershed: the proof-of-concept. Biodiversity. Université de Lyon, 2019. English. NNT: 2019LYSEN040 . tel-02447504

HAL Id: tel-02447504

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Numéro National de Thèse : 2019LYSEN040

THESE de DOCTORAT DE L'UNIVERSITE DE LYON
opérée par
l'Ecole Normale Supérieure de Lyon

Ecole Doctorale N°340
Biologie Moléculaire, Intégrative et Cellulaire (BMIC)

Spécialité de doctorat : Sciences environnementales
Discipline : Sciences de la Vie

Soutenue publiquement le 16/09/2019, par :

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**Implementing Ecosystem Natural Capital
Accounting Methodology to the Rhone
watershed: the proof-of-concept**

Application de la comptabilité écosystémique sur le bassin versant
du Rhône : validation du concept

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Acknowledgements/Agradecimientos

¡Que todo lo que respira alabe al SEÑOR!
Salmo 150:6

Mis más sinceros agradecimientos al Consejo Nacional de Ciencia y Tecnología (CONACyT) por la beca otorgada (225397/410110) para la realización de mis estudios de posgrado en Francia.

En Francia agradezco a la *École Normale Supérieure de Lyon* y a la *Ecole Doctorale de Biologie Moléculaire Intégrative et Cellulaire* donde he adquirido esta formación y recibí todo tipo de apoyo, administrativo, académico y personal. Hubo quienes hasta me brindaron su amistad. Agradezco en particular al *Institut Rhônalpin des Systèmes Complexes IXXI* y al *Laboratoire de Reproduction et Développement des Plantes RDP*.

A mi tutor Ioan Negruțiu por confiar en mí y aceptarme como su alumna para este proyecto, por su paciencia, sus consejos, su entusiasmo y por sus minuciosas revisiones. Sin él no hubiera sido posible emprender esta aventura.

A mi co-tutor Jean-Luis Weber por compartirme de su experiencia (¡que es mucha!), por su dedicación y por alentarme y aclarar mis dudas.

A François Golay y Tom Battin por la estancia en la *École Polytechnique Fédérale de Lausanne*.

A los miembros del Jurado:

Denis Couvet y Jean-Michel Salles por aceptar amablemente ser los revisores del manuscrito y por el tiempo dedicado para ello y por sus valiosos comentarios que ayudaron a enriquecerlo.

Anne Chin, Michel Trommèter y Olivier Hamant por aceptar amablemente formar parte del jurado, por su tiempo y sus observaciones.

A los miembros de los *Comités de Suivi de Thèse*:

Jean-Pascal Bassino, François Golay, Arezki Boudaoud, Luc Merchez por sus observaciones, comentarios y sugerencias para continuar con el trabajo de tesis y llevarlo a buen término.

A Hervé Parmentier y Romain Beauvais por la información brindada, por sus comentarios y observaciones.

Al grupo de *Evolution et Développement de la Fleur* del RDP por su disposición para escuchar mis presentaciones y por sus comentarios.

A mis compañeros y amigos informáticos, Nghi, Sarra, Mohammed, Marija, Sebastien y Jacob que tuvieron la paciencia y se tomaron el tiempo y la molestia de sentarse a mi lado para enseñarme y ayudarme a resolver muchos de los problemas que no habría podido resolver sin ellos.

A mi amigo Jacob que me ayudó de todas las formas y me acompañó en el proceso para la obtención de la beca.

A mis amigos que me han alentado y animado, Diana, Laura, Ursula, Aida, Marija, Ana.

A mi familia en Hillsong Lyon por ser la fuente donde puedo ir a beber del agua viva y el lugar de encuentro con mi Redentor. También agradezco por las amistades y oraciones, en especial por las chicas del *connect*.

A mis padres por todas sus lecciones, sobre todo por las de audacia, fortaleza y libertad para decidir. A mi mamá por animarme, por sus consejos y por todas las bendiciones que pone en mí. A mi papá ¡con dedicatoria especial! Sé que te haría feliz. Gracias por heredarme tu capacidad de sonreír en todo momento.

A mis hermanos por su resistencia. A Sandra por sacrificar de ella para beneficio de mí.

A Michel por ser mi inspiración para mejorarme cada día y ser un buen ejemplo para él y para invertir energía en hacer del mundo un lugar mejor.

A Rainer por atreverse a echarle chile a su vida, por su paciencia, disponibilidad, por ayudarme a organizar mis ideas y cálculos.

Abstract

How to measure the degradation of nature, its "health" condition, to determine the depreciation of its use, not recorded in the balance sheets of the nations? My thesis work on the "ecosystem accounting" of the Rhone river basin is an experimental first on a new methodology integrable with the still incomplete tools of the national accounts of the type GDP. Consuming ecological capital, our renewable resources, without amortizing means the creation of ecological debts.

The goal of ecosystem accounting is to produce a decision support tool to avoid the net degradation of ecosystems through science-informed public policies. The tool produces ecological balances based on geographical information and measured in physical terms, making it possible to perform modelizations and simulations in order to estimate the internalisation of externalities.

I have developed various synthetic indicators relating to the functions of ecosystems and their integrity, to calculate the ecological potential of the Rhone watershed and measuring ecosystem degradation, stability or enhancement. The accounting diagnosis is accompanied by a spatial description of the changes observed in order to better understand their relevance. The tool is likely to help the various territorial entities to position themselves on the management of their strategic renewable resources (water, soil, biomass, infrastructures and ecological services) in the context of a changing world order: major geopolitical issues in terms of security and food and energy sovereignty, in their link with public health.

Keywords: Ecosystem accounting, Carbon account, Water account, Ecosystem Infrastructure account, Biodiversity, Sustainable use, Ecosystem health, Ecosystem capability

Résumé

Comment mesurer la dégradation de la nature, son état de "santé", afin de déterminer l'amortissement de son utilisation, non-enregistré dans les bilans des nations? Mon travail de thèse sur la « comptabilité écosystémique » du bassin versant du Rhône est une première expérimentale sur une nouvelle méthodologie intégrable aux outils encore incomplets des comptabilités nationales de type PIB. Consommer du capital écologique, nos ressources renouvelables, sans l'amortir revient à créer des dettes écologiques.

L'objectif de la comptabilité écosystémique est de produire un outil d'aide à la décision permettant d'éviter la dégradation nette des écosystèmes par des politiques publiques informées par la science. L'outil produit des bilans écologiques basés sur des informations géographiques et mesurés en termes physiques, permettant d'effectuer des modélisations et simulations afin d'estimer l'internalisation des externalités.

J'ai élaboré divers indicateurs synthétiques relatifs aux fonctions des écosystèmes et de leur intégrité, pour calculer le potentiel écologique du bassin versant du Rhône et mesurer la dégradation, la stabilité ou l'amélioration de l'écosystème. Le diagnostic comptable est assortie d'une description spatialisée des changements observés afin de mieux saisir leur pertinence. L'outil est susceptible d'aider les différentes entités territoriales à se positionner sur la gestion de leurs ressources renouvelables stratégiques (eau, sols, biomasse, infrastructures et services écologiques) dans le contexte d'un ordre mondial en transformation: il s'agit des enjeux géopolitiques majeurs en matière de sécurité et souveraineté alimentaire et énergétique, dans leur lien avec la santé publique.

Mots-Clés : Comptabilité écosystémique, Compte de carbone écosystémique, Compte de la Ressource Écosystémique en Eau, Compte de l'Infrastructure Écosystémique, Biodiversité, Utilisation durable, Santé de l'écosystème, Capabilité écosystémique.

Table of Contents

Acknowledgements/Agradecimientos.....	i
Abstract.....	iii
Résumé.....	v
PhD. Objectives, Organization and Research strategy.....	1
Chapter 1 State-of-the-art in environmental evaluation since 1990.....	7
1.1 General introduction - article in “Dictionnaire juridique des transitions écologiques”.....	8
1.2 State-of-the-art in environmental evaluation.....	14
1.2.1. Background.....	14
1.2.2 Comparative analysis.....	17
1.2.3 Discussion.....	23
1.2.4 References.....	28
1.2.5 Supplementary documents	31
Outline of an international workshop on environmental evaluation 2020.....	33
Abstract.....	33
Rationale.....	33
The workshop.....	34
Pre-conference work	35
References.....	35
1.3 Planetary boundaries revisited, article in “Lancet Planetary Health”.....	37
Chapter 2 General introduction of the Ecosystem Natural Capital Accounts (ENCA).....	45
2.1 Introduction.....	45
2.2 Definition of statistic accounting units	53
2.3 Land cover stocks and flows.....	53
2.3.1 Methods.....	54
2.3.2 Results.....	56
2.3.3 Main issues.....	61
2.4 References.....	63
Chapter 3 Ecosystem Carbon Account.....	65
3.1 Introduction.....	65
3.2 Methods.....	68
3.2.1 Stocks.....	70
Bio-carbon in aboveground living biomass: Living trees tC ha-1.....	70
Bio-carbon in litter and deadwood	73
Bio-carbon in soil.....	74
3.2.2 Flows	76
Gross Primary Production (GPP).....	76
Net Primary Production (NPP).....	77
Heterotrophic respiration (RH).....	80

Net Ecosystem Production.....	80
3.2.3 Use.....	81
Agriculture harvested crops.....	81
Wood removals.....	82
Withdrawals of animal bio-carbon.....	83
3.2.4 Storing and retrieving spatial data.....	84
3.3 Results Carbon balance.....	86
3.3.1 Stocks.....	90
3.3.2 Flows.....	92
3.3.3 Use.....	94
3.4 Main issues.....	100
3.5 References.....	103
3.5.1 Data sources.....	104
Chapter 4 Ecosystem water account	107
Acknowledgement.....	107
4.1 Introduction - Water environmental accounts.....	108
4.2 Main features of ENCA Water and Rivers accounts.....	110
4.2.1 The system analysis behind the accounts.....	111
4.2.2 Definition and classification of statistical units for accounting	114
4.2.3 Hydrological units and other zonings of land.....	118
4.2.4 Units for soil water and groundwater accounting	120
4.3 The making of ENCA Water and Rivers accounts for the Rhône river catchment.....	123
4.3.1 The water accounting framework	123
4.3.2 The data requirements of ENCA Water and Rivers.....	126
4.4 Methods.....	134
Computation of rivers srkm values for compiling rivers quality account.....	134
Principles of river water quality accounting.....	134
Quality of water and condition of rivers: two different SRKM units.....	135
Methodological considerations for implementation.....	136
Implementation of the simplified method for calculating kmcn in the context of ENCA.....	137
4.5 Results - Rivers quality/ status account.....	138
4.5.1 Contextualizing the work.....	138
4.5.2. On the “Ecological Status”.....	139
4.5.3 Ecological Status data.....	140
4.5.4. Extraction of rivers status data to hydrological units (UZHYD).....	141
4.5.5 Calculation of average ecological status values by river class and by UZHYD.....	142
4.6 Issues, a synthesis of the data frame.....	151
Chapter 5 Ecosystem Infrastructure Account.....	155
5.1 Introduction.....	155
5.2 Methods.....	160
5.2.1 Basic Balances Parameters and Accessible Ecosystem Potential Indexes.....	160
Net Landscape Ecosystem Potential NLEP	160
Green Background Landscape Index GBLI	162
Landscape high nature conservation value index NATURILIS.....	163
Landscape connectivity index (meff).....	165
Net River Ecosystem Potential NREP.....	167
River Condition Potential RCP	168
River Natural Conservation Value NATRIV	169

5.2.2 Total Ecosystem Infrastructure Potential (TEIP)	170
5.2.3 Indexes of Intensity of use and Ecosystem Health	171
Index of Intensity of Use.....	171
Index of Ecosystem Health	172
5.3 Results and Discussion.....	176
5.3.1 Basic Balances and Accessible Ecosystem Potential.....	176
5.3.2 Total Ecosystem Infrastructure Potential	181
5.3.3 Indexes of Intensity of Use and Ecosystem Health.....	181
5.4 Main issues.....	187
5.5 References.....	189
Chapter 6 Integration of Accounts: The Ecosystem Capital Capability Account.....	191
6.1 Introduction.....	191
6.2 Methods.....	194
6.3 Results and Discussion.....	196
6.4 Main issues.....	204
6.5 References.....	205
Chapter 7 General Discussion.....	207
7.1 Proof-of-concept development.....	207
7.2 References	222
Chapter 8. Perspectives.....	223
8.1 Transborder ENCARHÔNE.....	223
8.2 France ENCA.....	225
8.3 Global ENCA.....	226
8.4 References	232

List of Figures and Tables

Chapter 1

Figure 1.1. Core indicators for Biodiversity, Ecosystem functions, Ecosystem goods and services, and Intensity of use.....	19
Figure 1.2. DPSIR model with main focus on ecosystem assessment and accounting frameworks.	22
Figure 1.3. Planetary Boundaries revisited (adapted from Steffen et al, 2015).....	40
Figure 1.4. Safe operating limits in high-dimensional risk spaces by representation of multi-dimensional threat levels (courtesy of Ralf Everaers).....	41
Table 1. 1. Categories of environmental evaluation including reference value, ecosystem services, and systems approaches.....	16

Chapter 2

Figure 2.1. Articulation and integration of ENCA components.....	50
Figure 2.2. Land-cover in the Rhone watershed.....	56
Figure 2.3. Land-cover change percentage from 2000 to 2012.....	57
Figure 2.4. Absolute net land-cover changes from 2000 to 2006.....	58
Figure 2.5. Drivers of land-cover changes from 2000 to 2006.....	59
Figure 2.6. Absolute net land-cover changes from 2006 to 2012.....	60
Figure 2.7. Drivers of land-cover changes from 2006 to 2012.....	60
Figure 2.8. Forest creation and management category 2000 to 2006 (a) and 2006 to 2012 (b).....	61

Chapter 3

Figure 3.1. Articulation and integration of ENCA components.	66
Figure 3.2. Bio-carbon in aboveground living biomass.....	72
Figure 3.3. Bio-carbon in litter and deadwood.....	74
Figure 3.4. Bio-carbon in soil for 0-30 cm layer. ESDAC 2013-2015 and INRA 2000-2009.	75
Figure 3.5. Estimations of bio-carbon in soil for 0-30 cm layer, 2000 (a), 2006 (b), 2012 (c).....	76
Figure 3.6. Gross Primary Production 2000 (a), 2006 (b), and 2012 (c).....	77
Figure 3.7. Net Primary Production DLR estimations for 2006.	78
Figure 3.8. Net Primary Production NASA estimations for 2000 (a), 2006 (b), and 2012 (c).....	79
Figure 3.9. Autotrophic respiration (RA) 2000 (a), 2006 (b), and 2012 (c).....	80
Figure 3.10. Agriculture harvested crops 2000 (a), 2006 (b), and 2012 (c).....	82
Figure 3.11. Wood removals at department level in 2000 (a), 2006 (b), and 2012 (c).....	83
Figure 3.12. Withdrawals of animal bio-carbon 2000 (a), 2006 (b), and 2012 (c).....	84
Figure 3.13. Dominant Land Cover Types (DLCT).....	85
Figure 3.14. Carbon stocks by integration of data on forest, deadwood, and soil in 2000 (a), 2006 (b), 2012 (c). (d) shows the change (%) from 2000 to 2012.....	91
Figure 3.15. Carbon flow. Net Ecosystem Production in 2000 (a), 2006 (b), and 2012 (c). (d) shows the change (%) from 2000 to 2012	93
Figure 3.16. Carbon use 2000 (a), 2006 (b), 2012 (c). (d) shows the percentage of change from 2000 to 2012. (e) illustrates the Dominant Land Cover Types in 2000.....	95
Figure 3.17. Carbon intensity of use 2000 (a), 2006 (b), and 2012 (c). (d) shows the change (%) from 2000 to 2012. (e) represents the carbon stocks in 2006.....	97

Figure 3.18. Carbon stocks, flow and use 2000 (a), 2006 (b), 2012 (c).....	99
Table 3.1. Carbon balance sheet 2000 (a), 2006 (b), and 2012 (c) compact versions.....	87

Chapter 4

Figure 4.1. Ecosystem Water Account in the ENCA framework.....	112
Figure 4.2. The water accounts model and main water categories.....	113
Figure 4.3. Rhône River watershed and catchment's main rivers.....	114
Figure 4.4. Statistic units at different scales and integration of data for the Rhone Water and River account (in French).....	115
Figure 4.5. Extract from the map of rivers with color codes for each class.....	116
Figure 4.6. The coding system of Sub-basins and ENCAT.....	117
Figure 4.7. UZHYD (black lines), ENCAT (white lines) and main rivers (red lines). UZHYD codes are those given by SANDRE. There is no tree coding.....	117
Figure 4.8. Intersection of DLCT and ENCAT (white lines) for producing SELU.....	119
Figure 4.9. Municipalities (in yellow) and UZHYD (in violet).....	120
Figure 4.10. Map of BD LISA's "couche affleurante" (green lines), ENCAT rivers (red lines), and ENCATs (white lines).....	122
Figure 4.11. Overlaps of shallow water tables ("nappes affleurantes").	122
Figure 4.12. Structure of ENCA-QSP Water Account (Weber, 2014).....	124
Figure 4.13. Rivers potential stocks in SRKM/km2 simplified methodology,.....	137
Figure 4.14: Assessment of status of surface water and groundwater according to the WFD. EEA report on European waters / Assessment of status and pressures 2018.....	139
Figure 4.15. Data source on water streams ecological status in 2009 (left) and 2015 (right).....	140
Figure 4.16. Principle of calculating quality weighted srkm by hydrological units.....	143
Figure 4.17. Change in ecological status by UZHYD (weighted srkm_R, %) 2009-2015,.....	145
Figure 4.18. Visual comparison between spatial distribution of agriculture panel (a) and changes of small rivers ecological status panel (b). Change in ecological status (weighted srkm_R, %) 2009-2015, All rivers by ENCAT, panel (c), and Small rivers (classes 4 to 6) by ENCAT, panel (d). ...	146
Figure 4.19 River Water Potential and River Condition Potential at SB_ENCA scale.....	147
Figure 4.20. Change in Rivers Condition Potential (weighted srkm_R, %) 2009-2015.....	148
Figure 4.21. River Condition Potential 2009, Large (a) and Small (b) Rivers.....	149
Figure 4.22. Gains or Losses of SB_ENCA Rivers Condition Potential 2009-2015 of Small Rivers (kmcn_R weighted by status index).....	150
Figure 4.23. Evolution of SB_ENCA River Condition Potential 2009-2015 in %.....	151
Table 4.1. Overview of data used for water accounts.....	127
Table 4.2. Extract from the file EtaEco_FRDR_MDORIV_09_15.csv.....	141
Table 4.3. SB_ENCA River Condition Potential 2009.....	148
Table 4.4. SB_ENCA River Condition Potential 2015.....	149
Table 4.5. SB_ENCA Gains or Losses of River Condition Potential 2009-2015.....	150

Chapter 5

Figure 5.1. Structure of ENCA-QSP Ecosystem Infrastructure Functional Services Account.....	156
Figure 5.2. Articulation and integration of ENCA components.	157
Figure 5.3. Net Landscape Ecosystem Potential for 2000 (a) and NLEP changes (%) from 201200 to 201200 (b)	161
Figure 5.4. Green Background Landscape Index 2000 (a), 2006 (b), 2012 (c).....	163

Figure 5.5. The Natural Conservation Value (NATURILIS)	165
Figure 5.6. Effective mesh size meff by SELU.	167
Figure 5.7. Net River Ecosystem Potential 2000 (a). (b) changes (%) from 2000 to 2012.....	168
Figure 5.8. River Condition Potential 2000 (a), 2006 (b), 2012 (c).....	169
Figure 5.9. Rivers from BD Carthage (left) and NATRIV index (right).....	170
Figure 5.10. Total Ecosystem Infrastructure Potential 2000 (a), 2006 (b), and 2012 (c).	171
Figure 5.11. Intensity of use 2000 (a), 2006 (b) and 2012 (c)	172
Figure 5.12. Biodiversity trend index for 2000 (a), 2006 (b) and 2012 (c).....	175
Figure 5.13. Green Background Landscape Index (a), NATURILIS (b), and CONECTIVITY (c) by Dominant Land Cover Type.....	179
Figure 5.14. Rivers Condition Potential (left) and River Natural Conservation Value (NATRIV) (right) by Dominant Land Cover Type.....	180
Figure 5.15. Total Ecosystem Infrastructure Potential by Dominant Land Cover Type.....	181
Figure 5.16. Index of Intensity of Use by Dominant Land Cover.....	182
Figure 5.17. Comparison of NLEP (a), NREP (b), TEIP (c), and Biodiversity index (d).	183
Figure 5.18. Comparative representations of Biodiversity evaluation.....	185
Table 5.1. Protected areas and their corresponding file codes (in French).....	163
Table 5.2. Protection scores.....	164
Table 5.3. Ecosystem Infrastructure Functional Services balance sheet compact version.	177

Chapter 6

Figure 6.1. Articulation and integration of ENCA components.....	192
Figure 6.2. Carbon Account Mean Internal Unit Value by SELU, 2000 (a), 2006 (b), and 2012 (c).	194
Figure 6.3. Water Account Mean Internal Unit Value by SELU, 2000 (a), 2006 (b), and 2012 (c)..	195
Figure 6.4. Infrastructure Account Mean Internal Unit Value by SELU, 2000 (a), 2006 (b), and 2012 (c).	195
Figure 6.5. Carbon unit value by Dominant Land Cover Type.....	197
Figure 6.6. Water unit value by Dominant Land Cover Type.....	198
Figure 6.7. Infrastructure unit value by Dominant Land Cover Type.....	199
Figure 6.8. ECU value by SELU, 2000 (a), 2006 (b), and 2012 (c).....	200
Figure 6.9. Ecosystem Capability Unit (ECU) by Dominant Land Cover Type.	200
Figure 6.10. Change in percentage of the Ecosystem Capability Unit by Dominant Land Cover Type.....	201
Figure 6.11. Geographic distribution of changes in ECU 2000 to 2012 (a), 2000 to 2006 (b), and 2006-2012 (c).....	202
Table 6.1. Ecosystem capability Unit value and trends during 2000 to 2012 period.....	199

Chapter 7

Figure 7.1. Conceptual model (Vitousek et al, 1997) on the growth of human population and the growth in the resource base used by humans.....	212
Figure 7.2. ENCA diagram indicating through a color code the articulation of relatively robust (green), satisfactory (blue), and fragile (red) components.....	213
Figure 7.3. ENCA, an integrator of environmental indicator systems and ecological transition priorities, namely land / soil, water, and biomass.....	214

Figure 7.4. Generating a unique modelling framework (Rugani et al 2016).....220

Table 7.1. Obstacles in the production of ENCA accounts.....216

Chapter 8

Table 8.1. Data requests made to OFEV/ FOEN.....223

PhD. Objectives, Organization and Research strategy

The context in which my PhD thesis has been conceived and achieved is the following.

A brief outlook of the political and economic determinants of the environmental crisis highlights the current “ecological degrowth”.

Environmental policies highly rely on natural, social, and information sciences which endeavor to supply decision makers with diagnosis, assessments, outlooks, quantitative norms, indicators, and integrated environmental statistical and accounting frameworks. Noticeable progress has been achieved in climate change modelling in support to global warming mitigation and to some extent and on a more *ad hoc* basis on sector policies, such as pollution abatement or the protection of high value nature areas. However, the ecological degrowth is at work, would it be global warming, resource depletion or the collapse of biodiversity. Social and economic trends are certainly the drivers of the process, but impacts are not the linear consequence of increasing needs. Action is possible to mitigate and even revert present trends, but it requires awareness of the issue, political willingness to act, and information in a format allowing its incorporation into public and private decision processes. Awareness is high now in all groups of the society; it has been raised by continuous communication of scientific messages to the public, using a variety of channels from scientific journals to TV reports. Policies seem to respond to these concerns, although results are often not what one could expect; policies are about trade-offs between a variety of demands, often contradictory, and short-term responses tend to prevail on longer term perspectives. A similar situation holds for the business sphere regarding its profits and the external costs, which have to be controlled, avoided, or simply ignored. As long as governments and businesses neglect the costs of their inaction or their negative impacts, no real change will take place. Indeed, the well established management tools that they use, the accounts, do not deliver the signals that foster a general and shared responsible behavior.

Most countries are not capable of adapting their resource use to the basic needs of their populations and maintaining life supporting natural systems on which societies depend. They do not incorporate ecosystem capital degradation in accounting standards. The overall degradation of natural systems resulting from economic and other activities is therefore equivalent to unpaid costs corresponding to ecological debts. The consequence is the accumulation of ecological debts over time, which creates

economic and political risks of which institutions as well as business are more and more aware of. There is an obvious need to both measure the degradation of nature to determine the depreciation / amortization of its use and record what has been measured in the balance sheets of the nations.

Measuring the wealth of nature, measuring the ecosystem capital has turned out to be a very much needed although complex operation. The primary purpose of my thesis is to assess the relevance and feasibility of natural capital accounts for incorporating ecosystem liability in economic decision making.

Prior to undertaking the research, the response to the situation by institutional and academic circles is analysed. A range of approaches, methodologies, and instruments of environmental evaluation and natural capital assessment have emerged in the last 20 years. They tend to integrate the ecological capital into economical and financial analysis. However, it is difficult to understand who is who in the field: the underlying goals of these environmental assessments are manifold and the corresponding tools developed to achieve those goals range from communication indicators to more operational metrics. There was so far no in depth comparative analysis of these methodologies, which makes hazardous any attempts of introducing them in decision-making. As long as such a contradictory conversation has not been engaged, and as long as some degree of a broad enough consensus has not been achieved on what tools to implement in order to sustain robust policies of strong sustainability, the clock will keep turning to the benefit of business as usual and of an accelerating accumulation of the global ecological debt.

My thesis work therefore consisted of three main objectives:

1. Make an inventory and perform a comparative analysis of current environmental and natural capital assessment approaches and methodologies.

This is developed in *Chapter 1*, that is structured in three parts. A note in French, corresponding to my first publication, reviews the environmental evaluation field in simple terms and could constitute an introduction to the field for decision-makers and other actors in the society. This is followed by the state-of-the-art analysis of the field and a sketch of a workshop necessary to mobilise the main stakeholders in environmental evaluation. The state-of-the-art analysis could serve as primer in engaging with the above mentioned debate.

Finally and for the sake of testing the conditions of such a debate, I explore the most recent concept and instrument in the field, the Planetary Boundaries, to conclude that the main components of this tool, also considered in most of the environmental evaluations panoply I analysed, correspond to three strategic primary resources: land, water, and biomass. This has made the object of a short publication meant to help clarify choices among the competing set of tools in environmental evaluation.

Our research strategy targeted the Ecosystem Natural Capital Accounts, or ENCA, as emphasized below.

2. Develop the argumentation for putting the ENCA guidelines and protocol to experimental test.

In the context of natural capital accounting experimentation, several tests have been or are being conducted worldwide. Most of them are case studies addressing one or the other aspect of ecosystems assets or services, without a concrete integration of the whole system.

The choice of ENCA for this PhD research project integrates the conclusions of Chapter 1 and stems from the following reasons:

- the methodology is an operational development of the volume on “Experimental Ecosystem Accounting” of the UN System of Economic Environmental Accounting (SEEA-EEA), building on previous work at the European Environment Agency (namely Land and Ecosystem Accounts, LEAC). The ENCA methodology has been published by the Convention on Biological Diversity (CBD);
- the methodology is directly compatible with existing instruments and statistics on which national accounts are produced;
- ENCA is a systemic approach and open protocol that is exhaustive by design (in terms of range and coverage of statistical data, their spatial representation and time series dynamics), while not prescriptive: it evaluates the state of the ecosystem capital over time and space;
- ENCA ambitiously targets no net ecosystem degradation (i.e. strong sustainability) by promoting the corresponding norms and standards for restoration and amortization costs.

The first part of *Chapter 2* develops the above considerations.

3. Develop a proof-of-concept tool of "ecosystem accounting" (ENCA) to evaluate the state of the ecosystem capital and make it actionable at the Rhône watershed scale.

The Rhône River ENCA project has been designed to address the whole public information system (scientific, statistical, and administrative) with the purpose of genuinely integrating data collected by the many actors involved in ecosystem monitoring and management (e.g. including data derived from Earth observation by satellite and other maps, meteorological and hydrological data, soil maps, biodiversity monitoring data, agriculture and forestry statistics, population censuses, or administrative registries). The work has implied assimilating more or less heterogeneous data, identifying gaps and filling them. Putting ENCA to work at the scale of a river basin has therefore been challenging, in particular for elaborating and organizing the methodology (I would rather say the methodologies) across the entire set of accounting entities, e.g. land, biomass-carbon, water, and ecosystem infrastructure. For each entity account, the collection and integration of various data sources has been a continuous challenge.

The thesis manuscript develops the proof-of-concept in Chapters 2-6. The headlines are as follows.

Chapter 2 starts with a terminology exercise aiming at clarifying the definition of statistical units in the environmental accounting field and in accounting-like approaches. On that basis, the ENCA accounting landscape is presented and illustrated by a general ENCA diagram, followed by considerations on land cover (**Chapter 2**) and rivers frame (**Chapter 4**): both constitute the matrix of the ENCA methodology.

Chapters 3, 4, and 5 develop the ENCA core accounts, namely the corresponding carbon, water and rivers, and ecosystem infrastructure components. Each account has methodological specificities that are highlighted.

Chapter 5 produces a series of intermediate-level indicators that target the ecosystem potential of the territory, such as the Net Landscape Ecosystem Potential and the Net River Ecosystem Potential that are combined to produce the Total Ecosystem Infrastructure Potential. Such synthetic indicators characterize the state of integrity of various land cover types and river ecosystems. Subsequently, the quantitative and more qualitative elements of these accounts are used in estimating indexes of

intensity of use of the specific resource and making the diagnosis of the state of the ecosystem, the ecosystem health.

Chapter 6 integrates the core account results and the intermediate-level indicators into a single synthetic macro-indicator of the ecological value, the Ecosystem Capital Capability (or overall Potential). The aggregation of the basic physical units of each account into a composite figure allows evaluating trends over time that inform on the level and rate of ecosystem capital degradation or enhancement.

The proof-of-concept I developed is the first experiment of this kind, at the meso scale, making broad use of a range of data made available by various agencies. I therefore attempt to critically discuss the obtained results (*Chapter 7*) in the perspective (*Chapter 8*) of the optimization of the accounting tool and its evaluation by potential users.

Achieving this ENCA data model proof-of-concept should allow reproducing, upgrading, and updating the accounts, ideally with the contribution of the agencies holding primary data. This has not yet been the case. For example, I discuss the important issue, raised on a different register in Chapter 1, of implementing ENCA on the entire Rhône river watershed. The Rhône river basin is a continuous ecological entity. Performing an integrated and exhaustive set of accounts of the natural resources within a geographic space with its own physical coherence enables a much better understanding and evaluation of actual internal transfers with little interference of arbitrary boundaries. As a matter of fact, that entity is disrupted by the political and administrative border between France and Switzerland. The proof-of-concept tool reported here is limited at present to the French part of the basin and a few variables for Switzerland reported to the European Environment Agency or easily available in a compatible format. It happens that Switzerland has opted for the planetary boundary approach in the evaluation of its natural capital. This has implications on the generation of the current Swiss data system for environmental evaluation. A future ENCA test for Switzerland would certainly benefit of this new data resource. In addition, it would allow deepening the reflection on information tools initiated in this proof-of-concept.

In conclusion, I will try to argue why the ENCA approach and my work are original contributions to the field, wherefrom ENCA borrows and shares methodological elements with other approaches, and to what extent ENCA develops the experimental guidance of the UN System of Environmental-

PhD. Objectives, Organization and Research strategy

Economic Accounting (SEEA) and other contributing agendas into an actionable protocol having the capacity to put ecosystem and biodiversity issues on par with climate change on the international policy agenda.

ENCA overlaps with other existing frameworks targeting more specific issues such as ecological footprint, water footprint, carbon balances, material flow accounts, human appropriation of primary biomass production, boundary conditions to the use of critical resources, assessment of ecosystem services, and various other indicator sets. While they extend beyond the scope of ENCA, they all have an entry point to ENCA (see General Discussion, *Chapter 7*). Mutual benefits of that are data sharing and cross-analyses. It is particularly important considering the interpretation of each indicator in the light of ecosystem degradation measured by ENCA.

ENCA was designed as an integrated framework focussing on the measurement of ecosystems capability of reproducing themselves and providing their services continuously, and of ecosystem degradation or enhancement when and where it happens. Ecosystem capability is measured for all ecosystems in a unit-equivalent and aggregated indicator which merges quantitative and qualitative indexes computed in the accounting framework. Measuring ecosystem degradation allows to assess public and private actors' liability and to propose new policies that consider restoration costs in the perspective of reducing ecological debts and ultimately incorporating ecosystem capital depreciation into accounting standards.

My work on the Rhône river watershed is the first attempt to prove the relevance and feasibility of ENCA at a sub national scale. It has been carried out by making use of data and statistics presently accessible from public agencies. The ENCA data model is guidance for assimilating these very heterogeneous data sources into an interconnected set of grids. However, while accounting techniques based on cross-checking have allowed assessing data consistency and proposing moderate adjustments, they have highlighted data gaps which have had to be closed by estimations. Under similar conditions, the proof of ENCA feasibility for 2,225 Socio-Ecological Landscape Units of the Rhône river watershed is reported here, including the compilation of their Total Ecosystem Capability. This having been said, moving from a pilot to an operational information system will require systematically fixing data gaps with the support from various agencies and scientific bodies presently collecting these data within the frame of a coherent and ambitious public policy of ecosystem capital accounting.

Chapter 1 State-of-the-art in environmental evaluation since 1990

Outline

The chapter consists of three parts. The first is a note in French on Environmental Assessments published in the *Dictionnaire juridique des transitions écologiques*, introduced by an abstract in English. The second part is the state of the art analysis called “Decision making based on ecosystem value” and supplementary files, among which a series of Tables (to be accessed through a link) in which the main approaches and the corresponding methodologies in environmental evaluation are compared and the outline of an international workshop on environmental assessment that draws its substance from the state of the art comparative analysis. This is followed by a short publication analysing the Planetary Boundaries concept and the hidden relevance of three strategic primary resources: land, water, and biomass. Taken together, these distinct introductory elements prepare the ground for developing a proof-of-concept tool of "ecosystem accounting" (ENCA) by putting the ENCA guidelines to experimental test.

1.1 General introduction - article in “Dictionnaire juridique des transitions écologiques”

Environmental evaluations

Evaluating the dynamics and balance between extraction and use and the regenerative capacity of natural capital and its contribution to the economy and the well-being of the people are of growing importance for decision-making and governance. The frameworks that are produced and tested to evaluate nature are ecological indicators and footprints, monitoring and evaluation of ecosystem services, and environmental accounting. *Indicators and footprints.* Indicators are constantly appearing and are mainly targeted to counterbalance the Gross Domestic Product (GDP) metric. Environmental footprint indicators aggregate ecological impacts and / or the use and efficiency of resources (such as materials, water, soils, greenhouse gas (GHG) emissions). *The ecosystem services* approach draws on its apparent potential to facilitate decision-making for more efficient use of resources, to avoid unintended negative impacts, and to find trade-offs between the divergent interests of the various actors. *Ecosystem accounting.* Global and systemic accounting of natural / ecosystem capital should at the same time make it possible to estimate the reproduction, the supply, the use, the consumption, the accumulation and the trade of the natural resources while taking into account the remunerations, the profits, taxes and subsidies, financial flows and assets. It should provide a solid statistical basis for calculating ecosystem capital consumption and its translation into depreciation not recorded in conventional accounting, which corresponds to ecological debts generated along the entire process. On what basis will the choices of operational instruments be made? A distinction should be made between those whose virtues are more pedagogical (the ecological footprint, for example) and those with real decision-making abilities. They need to be tested and validated in various socio-cultural contexts to introduce into current accounting practices the full depreciation of the natural capital used. It also remains to design a solid consistency at different scales of political organization.

Évaluations environnementales

Les cycles et les fonctions des écosystèmes génèrent des stocks et des flux d'énergie et d'information, à partir desquels les humains obtiennent des ressources et autres services sous forme de biomasse, d'eau accessible etc. Concevoir l'écosystème comme une forme renouvelable de capital naturel conduit à ce que les flux de biens et services d'origine écosystémique soient traités convenablement dans l'évaluation des résultats de l'économie. Ce n'est pas le cas car le coût

complet du renouvellement des actifs naturels n'est pas pris en compte (BARTELMUS, 2009) et donc la nécessité d'amortir la consommation de capital naturel renouvelable est ignorée.

Évaluer la dynamique et l'équilibre entre extraction-utilisation et la capacité de régénération de ce capital et sa contribution à l'économie et au bien-être des populations revêtent une importance croissante pour la prise de décision et la gouvernance. Une gouvernance des systèmes socio-écologiques devrait évoluer vers des formes adaptatives et davantage participatives.

Mesurer le capital naturel. Les États restent incapables d'adapter les ressources dont ils disposent aux besoins vitaux de leurs populations. Parmi les raisons, il y a les règles internationales sur les investissements et le commerce (COLLART DUTILLEUL, 2012). Plus généralement, gouvernements et entreprises ne tiennent pas de comptabilité environnementale et sont incapables de mesurer les coûts de la dégradation de la nature engendrés par son exploitation. Ceci tient notamment au fait que l'écosystème est composé d'actifs exploitables qui ont des fonctions sociales qui vont au-delà des droits de propriété de leurs titulaires. Consommer du capital écologique sans l'amortir revient à créer des dettes écologiques qui sont transmises aux générations présentes et futures, mais également aux pays dont nous importons des biens produits non-durablement. Cette culture systémique des déficits révèle un système économique subventionné (DASGUPTA, 2010) et une gestion des ressources et de l'aménagement des territoires problématiques.

Faire une évaluation de la nature comme « capital » qui ne soit pas uniquement en termes monétaires (par exemple la valeur intrinsèque de la nature, les coûts de restauration d'un écosystème endommagé ou le « prix » d'une espèce) suppose de définir clairement des limites entre ce à quoi on peut ou on ne peut pas donner un prix (VANOLI, 2005) et ainsi soumettre ces catégories ou non aux règles du marché, et selon quelles modalités. À présent, les instruments qui sont produits et expérimentés sont des indicateurs et des empreintes écologiques, le monitoring et l'évaluation des services écologiques, et la comptabilité environnementale.

Indicateurs et empreintes. Des indicateurs apparaissent en permanence et sont principalement ciblés pour contrebalancer la métrique du Produit intérieur Brut (PIB). Les indicateurs de type empreinte environnementale agrègent des impacts écologiques et/ou l'utilisation et l'efficacité des ressources (comme les matériaux, l'eau, les sols, les émissions de Gaz à effet de serre (GES). Le « E-RISC report » (2012) décrit les méthodes et la métrique permettant de quantifier la consommation des ressources renouvelables en particulier et d'évaluer les risques correspondants traduits en termes de dette écologique. L'indicateur principal proposé est l'empreinte écologique, mais sa fiabilité a été mise en question (BLOMQVIST *et al*, 2013).

Les services écosystémiques. Le concept est basé sur un éclatement du système écosystémique en services distincts. Les données sur l'utilisation des stocks et des flux (physiques et monétaires) dans les écosystèmes servent à déterminer des biens et des services issus de la nature. Le programme « MAES » de la Commission européenne se focalise sur la cartographie et l'évaluation physique des services. Certaines méthodologies (privilégiées par exemple par le programme WAVES de la Banque Mondiale) tendent à monétiser la nature et à se concentrer sur des actifs naturels à potentiel économique (MAZZA *et al*, 2013). Dans son programme national d'évaluation des écosystèmes sur la base d'un « produit écologique brut », la Chine tente de stimuler les investissements de conservation du capital naturel dans les régions qui génèrent le plus de services et pour rendre son agriculture plus durable (OUYANG *et al*, 2016). L'approche par les services attire par son potentiel apparent à faciliter la prise de décision pour une utilisation plus efficace des ressources, pour éviter des impacts négatifs non-intentionnels, pour trouver des compromis entre les intérêts divergents des divers acteurs.

Comptabilité écosystémique. Une comptabilité globale et systémique du capital naturel devrait tout à la fois permettre d'estimer la reproduction, l'approvisionnement, l'utilisation, la consommation, l'accumulation et le commerce des ressources naturelles tout en tenant compte des rémunérations, des profits, des taxes et des subventions, des flux financiers et des actifs. Elle devrait en particulier fournir une base statistique solide pour calculer la consommation de capital écosystémique et sa traduction en amortissement non enregistrés dans les comptabilités conventionnelles, qui correspondent aux dettes écologiques engendrées le long de l'ensemble du processus. En d'autres mots, capter les quantités et les flux de stocks accessibles sans dégradation des ressources et de l'environnement. Les avantages des systèmes de comptabilité environnementale sont considérables (MAZZA *et al*, 2013) : Elle est appelée d'une part à compléter des comptabilités nationales et d'entreprises établies par la mise en place de bilans écologiques en unités physiques et en monnaie. Cette démarche repose sur l'approche systémique développée par la Commission statistique des Nations Unies et l'Agence Européenne pour l'Environnement et est synthétisée dans la méthodologie publiée par la Convention de l'ONU sur la Diversité Biologique (WEBER, 2014). Elle constitue une approche intégrée des structures et fonctions des écosystèmes évaluées en termes d'étendue, de productivité et de facteurs de résilience et de leur utilisation économique et sociale (divers services). La comptabilité écosystémique génère une infrastructure statistique cohérente au service des systèmes de décision à différentes échelles géographiques et économiques.

D'autre part, les comptes écosystémiques sont un élargissement des budgets « équivalents CO₂ » visant à internaliser la dégradation du climat dans les processus de décision des États et des entreprises. Ces budgets mesurent des crédits et des débits sur lesquels se fondent des normes réglementaires ou fiscales et des mécanismes de marché basés sur un prix du carbone destiné à en réduire l'utilisation. Les comptes écosystémiques sont plus complexes. Ils utilisent d'une part les bases de données de la couverture des sols, de la biomasse (bio-carbone), de l'eau et divers indicateurs de haut niveau relatifs aux fonctions des écosystèmes (comme l'intégrité des infrastructures vertes et bleues) et d'autre part les statistiques socio-économiques relatives aux prélèvements de ressources matérielles et aux usages intangibles. Ils agrègent ces états en une mesure composite de capabilité, similaire dans sa conception à l'« équivalent CO₂ » des budgets du GIEC. Les unités de capabilité écosystémique permettent de calculer sur une base annuelle l'amortissement de capital naturel et de l'incorporer dans les standards comptables. Les comptes écosystémiques physiques peuvent être valorisés en monnaie, en particulier en ce qui concerne les coûts de restauration ou d'évitement de la dégradation des écosystèmes. En cela, ils diffèrent de la comptabilité détaillée de la valeur d'usage des services écosystémiques, tel que proposée par la Banque mondiale.

Sur quelles bases seront effectués les choix des instruments opérationnels ? Les évaluations environnementales sont actuellement assez précises lorsqu'on se réfère à des variables comme les flux de matériaux, les émissions dans l'atmosphère, les stocks et flux d'énergie et d'eau (en termes physiques et monétaires), les taxes, les dépenses couvrant des risques et des subsides. Les écosystèmes et la biodiversité restent beaucoup plus difficiles à mesurer (MAZZA *et al*, 2013). Le développement rapide des systèmes de monitoring par satellite et *in situ* et des systèmes de traitement des données permettent de commencer à mieux suivre les dynamiques des écosystèmes.

À terme, il s'agit de se doter de capacités de recherche indépendantes permettant de comparer des instruments d'évaluation du capital naturel pour pouvoir distinguer entre ceux dont les vertus sont plutôt pédagogiques (l'empreinte écologique, par exemple) et ceux possédant des capacités réelles d'aide à la décision. L'essentiel est de « *dégager les finalités de société afin de s'assurer de la qualité intrinsèque des biens communs naturels avec des indicateurs pertinents, essentiellement non-monétaires* » (GADREY et LALUCQ, 2015).

Selon GADREY et LALUCQ, « *l'évaluation monétaire de la nature (les paiements pour services écosystémiques, les marchés dérégulés de permis de droit à polluer, les banques de compensation etc.) ne peut en aucune façon constituer l'outil dominant d'une politique de*

préservation de la nature », car cela reviendrait à accepter que l'on protège la nature pour autant que cela rapporte de l'argent.

Dans la réalité, le paysage des instruments est en train de changer d'une manière non-coordonnée. Le Règlement communautaire n° 691/2011 prévoit l'application des comptes économiques environnementaux par les pays membres et la stratégie Biodiversité de l'UE promeut la cartographie et l'évaluation des écosystèmes et des services associés en prévision de la mise en œuvre des systèmes de comptes et de *reporting* en 2020.

Un cas d'école est le « *Resource Management Act* » de la Nouvelle Zélande (1991). Cette loi suprême rassemble et articule plus de 50 lois sectorielles de l'environnement en un seul corpus qui ambitionne de construire un modèle de développement prenant en compte la reconnaissance de la valeur intrinsèque des écosystèmes, de la nécessité de préserver et de protéger l'héritage et les droits coutumiers, d'évaluer les impacts des activités humaines sur l'environnement afin de les éviter ou d'y remédier. Toutefois son application est contrariée par des contextes législatifs, politiques et économiques, comme c'est le cas de la politique agricole du pays sous la pression des forces du marché mondial.

Nous disposons donc aujourd'hui d'un arsenal d'instruments politiques et juridiques, mais aussi d'outils techniques d'évaluation, qui demandent à être expérimentés et validés dans des contextes socioculturels divers pour introduire dans les pratiques courantes l'amortissement à part entière du capital naturel exploité. Reste aussi à imaginer une solide mise en cohérence à différentes échelles d'organisation politique.

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Jean-Louis WEBER

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Liens avec d'autres entrées du dictionnaire :

Capital naturel ; Comptabilité environnementale

1.2 State-of-the-art in environmental evaluation

To introduce and position the PhD work and in the absence of an exhaustive and up-to-date methodological analysis of the field, I generated a comparative study on environmental evaluation approaches and tools. This is meant to become a review paper on the subject.

On terminology. The literature makes use of environmental evaluation, natural capital evaluation, and ecosystem capital evaluation to address the necessity of building new approaches and instruments to assist decision-making. The various indicators, statistical and accounting frameworks have similarities as they cover the same realm, but differences in their approaches, formats and purposes. I will come back to that in **Chapter 2**.

Decision-making based on ecosystem value

1.2.1. Background

Most States are not capable of adapting their resources to the basic needs of their populations (FAO, 2016; Mazoyer et al, 2008) and maintaining of life-supporting natural systems for reasons that are, among others, linked to the fragmentation of policies and globalization of markets, the international laws pertaining to international investments and trade (Collart Dutilleul, 2012; Honet and Negrutiu, 2012), the expanding unsustainable food systems (UNEP 2016; De Schutter 2017), and denial in taking nature into account in decision-making by the wide use of incomplete and / or misleading indicators of production, income, consumption, saving, investment, and debts (Weber, 2018).

Here we focus on the “natural capital-time-justice” frame. The degradation of the life-supporting capacity of natural systems, that directly and indirectly provide livelihoods, results in the depletion of material resources and in the alteration of the structures, functions, and cycles that allow their renewal. Such functions are public goods that are consumed as “externalities”. Ignoring degradation is due to the fact that it largely escapes the economic calculation on which decisions are made: externalized costs are not integrated in the price of products that incorporate ecosystem functions and services. GDP is a relatively objective measure of economic transactions with little concerns about social and ecological consequences. GDP is only recording the monetary value of ecosystem services embedded in derived products in view of benefits (i.e. stocks of natural resources with economic value, such as forests, timber, and fish) and – to various extents, the environmental protection activities and expenditures for air, soil, water, biodiversity, and landscapes (Dasgupta, 2010; Weber, 2018). While in GDP the consumption of fixed capital is subtracted in order to maintain industrial and infrastructure assets, the cost of natural capital regeneration to provide

functional ecosystem services - such as water quality, soil fertility, fish stock regeneration, landscape-related elements of quality of life, for example, is ignored (Weber, 2018).

The overall degradation of natural systems resulting from economic activities is therefore equivalent to unpaid costs corresponding to ecological debts. These debts are virtual as long as they are not measured and recorded in order to be offset. The consequence is a deterioration of ecological balance with the accumulation of ecological debts over time representing economic and political risks (such as risks taken by suppliers and financiers that support activities harmful to the ecosystem) (Caldicott et al, 2013; WEF 2015;).

Hence, the strategic interest in the evaluation of the state of resources and the consumption of ecosystem capital through the lens of the ecosystem health (Rapport et al, 1998), namely the ecosystems' infrastructure integrity, diversity, and resilience. That notion has spurred a series of international frameworks such as the Millenium Ecosystem Assessment (2005) and The Economics of Ecosystem and Biodiversity, UN Statistical Commission (2012), the IPCC guidelines on carbon emissions (2006), CBD Convention on Biological Diversity strategies, OECD reports on green growth and environmental, social, and governance (ESG) factors (reviewed by Weber, 2018) etc. They supported during the last 20 years the emergence of a broad range of approaches, methodologies, and instruments of environmental evaluation that aim integrating the environment and natural resources into economical national accounting frameworks.

The goals behind these approaches are manifold, aiming at (**Box 1.1**):

Box 1.1 Environmental valuation approaches goals and outcomes

Goals	Outcomes
1. Providing information for decision-makers on natural resource supply, stocks, and use.	Allowing to assess resource depletion and resource-use efficiency.
2. Integrating the environment and natural resources more completely within economic decision making.	Valuation of services supplied by the ecosystems to be subtracted from the created income.
3. Assessing the wealth of nations by an inclusive approach to Capital made of produced, human, social, and natural capital.	Natural capital evaluation allows adjusting conventional macroeconomic indicators to account for the depletion of resources, assets degradation due to pollution, and incorporation of the value of non-marketed flows of goods and services derived from nature.
4. Accounting for environment at the micro-economic level (corporate, local government, projects).	For example, environmental assessments best describe the resource landscape in which corporate operates.

5. Measuring the ecological value of ecosystems from physical assessments to compute ecosystem degradation.	Offsetting nature degradation and incorporating ecosystem capital depreciation in accounting standards.
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This study is a comparative analysis of current environmental evaluation methodologies (**Table 1.1**) covering distinct approaches in the field (see also earlier reports of Mazza et al, 2013; UNEP, 2014; Weber, 2018). We are aiming at distinguishing tools with pedagogic value from tools effective in decision-making. The latter are essential in making trade-offs / arbitrating between competing or conflicting interests of socio-economic, institutional, or territorial entities, in a logic of anticipation and *in fine* of securing the ecosystem capital. We analyse the indicators from an ecosystem perspective by considering their correspondence to the ecosystem health concept (Rapport et al, 1998; Müller and Burkhard, 2010). We propose upgrading existing methodologies, the objective being to achieve no net degradation of the ecosystem capital in real life through public policies aiming at strong sustainability.

Indicators, Footprint and Boundaries	Land Cover/Use
Environmental indicators (OECD, 1990)	Land Cover Accounts (LEAC, 2006)
Ecological Footprint (EF, 1997)	Economics of Land Degradation (ELD, 2011)
Planetary boundaries (Steffen, 2015)	Global Land Project (GLP, 2015)
Ecosystem Services Assessment	Environmental Accounting
Millennium Ecosystem Assessment (MA, 2000)	System of Environmental and Economic Accounting (SEEA 1993, 2003, 2012)
The Economics of Ecosystem and Biodiversity (TEEB, 2008)	
Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2012)	Wealth Accounting and the Evaluation of Ecosystem Services (WAVES, 2010)
Mainstreaming ecosystem services into EU policy (MAES, 2013)	Canada's Measuring Ecosystem Goods and Services (MEGS, 2012)
UK National Ecosystem Assessment (UKNEA, 2011)	Australian Environmental-Economic Accounts (AEEA, 2014)
China's National Ecosystem Assessment (CEA, 2016)	Ecosystem Natural Capital Accounts (ENCA-QSP, 2014)

Table 1.1. Categories of environmental evaluation including reference value, ecosystem services, and systems approaches. The four designated categories are (1) indicators - ecological footprints - boundaries, (2) land cover and use, (3) ecosystem services, and (4) environmental accounting. Most of them are extensions of carbon budgets according to IPCC work (2006) with variations on other geo-physico-chemical and biological parameters.

Several methodologies that integrate biomass state variables target the Human Appropriation of Net Primary Productivity (Haberl et al, 2007). Technically, evaluations are established in physical and/or monetary units.

1.2.2 Comparative analysis

Supplementary Table 1.1¹ details the items and references presented in **Table 1.1** by organizing the corresponding information according to the following criteria (13 out of the total of 20):

Objective(s) / goal(s); definition of terms; context and strategy of research; analytical approach; methodology and data sources; indicators and relevance to DPSIR; accounting tables (accounting in physical and / or monetary units); strength versus weaknesses / limitations; policy relevance (aid to decision-making); implementation as bottom-up versus top-down approaches; validation through case studies, national to global reporting; periodicity; peer-review evaluation to distinguish methodologies that comply to a science-prone procedure.

The main trends are highlighted below.

The Objectives criterion makes it possible to see how the analysed methodologies match the goals listed in the Introduction (see **Box 1.1**). The majority (EF, 1997; LEAC, 2006; ELD, 2011; GLP, 2015; MA, 2000; TEEB, 2008; IPBES 2012; MAES, 2013; UKNEA, 2011; CEA, 2016; SEEA, 2013; WAVES, 2010; MEGS 2012; AEEA, 2014) belongs to goal 1, namely “providing information for decision-makers on natural resource supply, stocks, and use”. They have been therefore designed to target weak sustainability.

By intersecting the criteria Strategy, Analytical approach, and Strengths and Limitations we note that most current methodologies consist of guidelines and analytical and experimental framework building (OECD, 1990; LEAC, 2006; ELD, 2011; TEEB, 2008; MAES, 2013; SEEA, 2013; WAVES, 2010; ENCA, 2014). For example, as part of the SEEA, the UN Statistical Division, the United Nations Environment Program, the Secretariat of the Convention on Biological Diversity, and the European Union have launched in 2007 the project “Natural Capital Accounting and Valuation of Ecosystem Services” (NCAVES). It aims to assist the five participating partner countries, namely Brazil, China, India, Mexico, and South Africa, to advance the knowledge agenda on environmental-economic accounting, in particular ecosystem accounting.

By comparison, other methodologies were dedicated to in context evaluation of chosen geographical entities, namely:

¹ https://drive.google.com/open?id=1yWuRStKnltqS7XtDp2n_dE1v_DpSI7A3

Chapter 1 State-of-the-art in environmental evaluation since 1990

- (1) Case studies, designed and adapted to local contexts and stakeholders (ELD, 2001; TEEB, 2008; WAVES, 2010; IPBES, 2012) and the inherent difficulty that systematic and meaningful comparisons are unlikely;
- (2) National level evaluations that are in the calibration stage, and - with the exception of Australia, have not (yet) a defined periodicity;
- (3) The Ecological Footprint is the only tool that has systematically been deployed at different territorial scales.

There is a tendency across the analysed methodologies (ELD, 2001; TEEB, 2008; SEEA, 1993; WAVES, 2010; UKNEA, 2011) to adopt monetary valuations as solution for aggregating distinct (sub)categories of ecosystem capital.

More generally, our analysis shows that progress and faster diffusion of environmental accounting frameworks are hindered by the lack of sufficiently robust and spatially explicit data (see also below). Therefore, no comprehensive comparisons between the panoply of current methodologies can be performed for an in depth understanding of their respective strengths and limitations. However, for the sake of enabling appropriate choices among the current tools for decision-making in the short term, experimental work should be considered on test-territories where data sources are sufficiently exhaustive and robust.

In addition, most of the current methodologies have not been scrutinized through the peer-review process (only EF, GLP, IPBES, MAES, SEEA and CEA have; Blomqvist et al, 2013; Verburg et al, 2015; Díaz et al, 2015; Maes et al, 2016; Bartelmus, 2009; Ouyang et al, 2016). One ELD case study (Favretto et al, 2016) and the water accounting in Australia (Vardon et al, 2017) can also be cited. This further puts a burden on the choice of best fit decision-making tools.

In what follows we focus the analysis on the ecosystem-to-natural capital dimension and relationship. The indicators related to the analysis of biodiversity, ecosystem functions, ecosystem services, natural capital-resources and their intensity of use as reported in the analysed methodologies are detailed in **Supplementary Table 1.2**. Their graphic representation for each of the four methodological categories is presented in **Figure 1.1**.

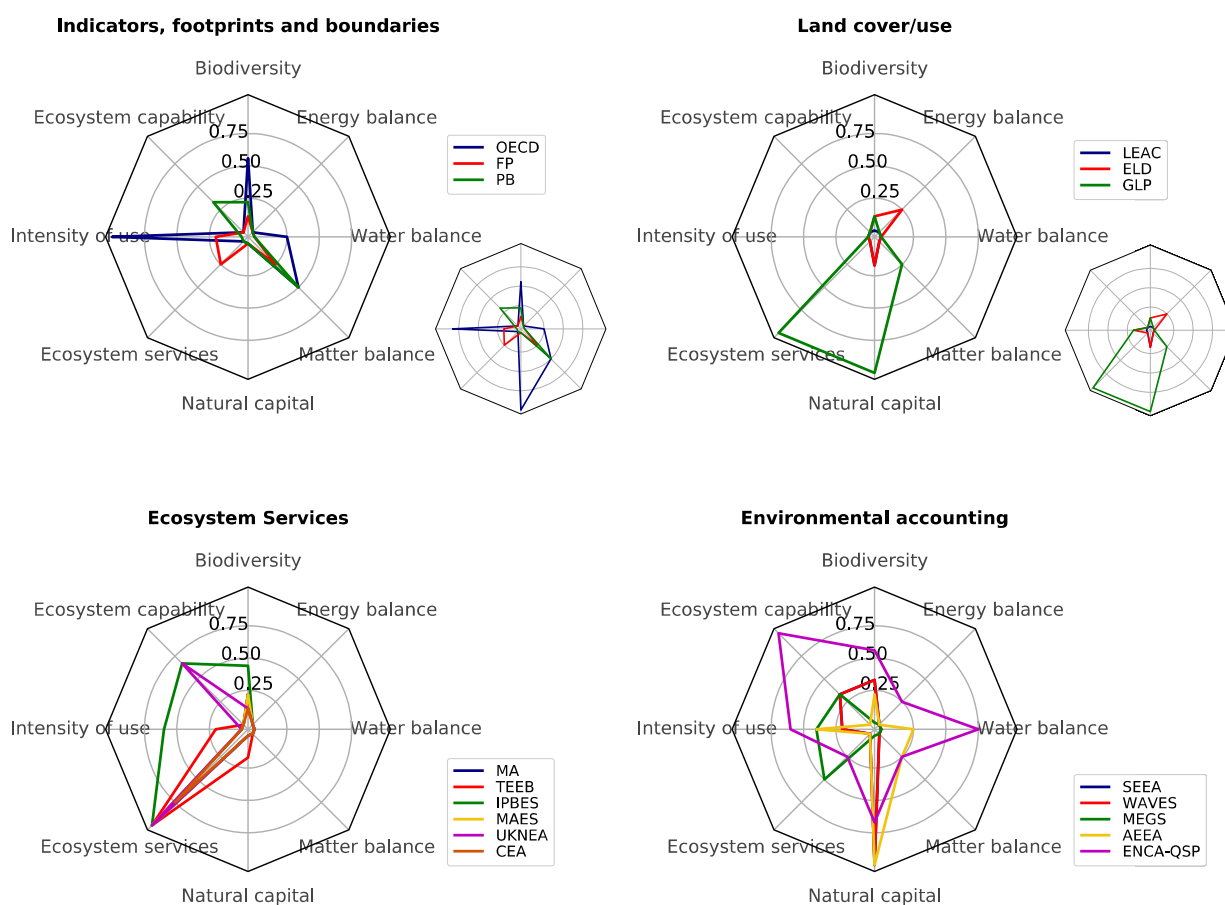


Figure 1.1. Core indicators for Biodiversity, Ecosystem functions, Ecosystem goods and services, and Intensity of use. The scores I mapped were calculated as a proportion of the indicators found in the frameworks. The further away from the center a line crosses a specific axis, the higher the attributed value in a given methodology to the particular indicator. The two insert diagrams are our extrapolation of scores based on the assumption that natural capital (OECD) and intensity of use (GLP) are implicitly considered and evaluated in their respective indicator system.

Concerning Biodiversity, and despite the general adoption of the Convention on Biological Diversity definition (see **Suppl. Table 1.1**), it is measured as a structural feature of ecosystems (PB, 2015; GLP, 2015; MA 2000; TEEB, 2008; IPBES; AEEA, 2013; ENCA, 2014) or as resource (OECD, 2001; WAVES, 2010; SEEA, 1993; AEEA, 2014; CEA, 2016) or both (MAES, 2013; UKNEA, 2011). In the Ecological Footprint approach, the footprint itself is considered as a biodiversity indicator. In brief, biodiversity is systematically but variably accounted for in the analysed methodologies. The indicators we scored agree with eight of the nine Essential Biodiversity Variables (EBV; Pereira et al, 2013), only population diversity is not represented by

any indicator. The result shows that OECD, IPBES, and ENCA account more readily for the chosen EBVs.

The assessment of Ecosystem functions, a proxy for ecosystem health or integrity measured as energy, water, and matter balance, turns out to be rarely considered in the analysed methodologies, with the notable exception of ENCA. Of note, the matter balance is more frequently evaluated than water and energy balance.

The matter balance takes us to Natural capital and Ecosystem services, i.e. the benefits humans take from nature. The two indicators are almost systematically employed in all the analysed categories.

The intensity of resource use (essentially water, forest, fish, and energy) is as a matter of concern in all categories, except that the Land Use group does not seem to cover this dimension. In ENCA, the indicator is calculated as the quotient of the resource use and productivity, and represents sustainable use (Weber, 2014).

Taken together, the Indicators-Footprint group appears resource- and use value oriented, the set of indicators being rather disjointed. Ecosystem Service and Land Use approaches follow a similar trend. Of note, the analysis of the land group shows that the resource has gained momentum in the last decade. Both categories perform poorly on indicators for ecosystem functions and cycles (see the right half of the diagram), i.e. for measuring ecosystem integrity, degradation, and resilience. The Environmental Accounting group has a broader coverage of the diagram, with ENCA best integrating ecosystem functions, both quantitatively and qualitatively. This correlates well with its high score estimating the ecosystem capability (potential). In consequence, this group is expected to best account for the intrinsic value of ecosystems.

Finally, policy relevance and data infrastructure were evaluated in terms of bottom-up versus top-down implementation practice. Production of and access to data stand out as a critical issue. Several frameworks and guidelines have been proposed by international expert bodies to match statistical contexts at regional, national, and/or local levels. However, this is rarely the case. Frequently results are produced from aggregated data, usually derived from national statistics or metadata. Alternatively, surrogate modelling and data extrapolation are promoted for some tools in order to palliate the lack of data (Natural Capital Coalition, 2019).

Two experimental frameworks have ensued. On the one hand, case studies – steadily multiplying and so useful they might be, are essentially based on circumstantial information collection. On the other hand, certain States have initiated top-down implementations (Canada, Australia) aiming at monitoring environmental stocks and flows that are relevant to country's economy and society.

These assessments are therefore designed with policy relevance in mind (certain TEEB study cases, IPBES, 2012; MEGS, 2013). Of note, in Australia the assessments are integrated in the National Accounting System (AEEA, 2013). They ensure developing at governance level data policies and frames, an advantage in terms of building data infrastructure, standards, and classifications as part of national balance sheets, and ultimately to ensure the necessary periodicity of the process.

The results of the bottom-up assessments have also been exploited in policy-making. For example, in China they served to extend protected areas in the country and to identify sensitive areas for protection when designing road projects (CEA, 2016). At local level, the *TEEBcase Mexico* served for payments for ecosystem services to rural land managers by selling carbon certificates.

Taken together and despite certain achievements, a recent survey (Recuero Virto et al, 2018) concluded that “there is very little use of natural capital accounts for public policy decisions and, more so, in developing countries”. The main barriers were considered to be the lack of political support, institutional obstacles, and insufficient data availability and cooperation.

Furthering the comparative methodological analysis should take into consideration two more aspects:

(1) Categorize the reported methodologies in the light of the DPSIR analysis by a refined dissection of how the indicators proposed by the model match those actually used. The DPSIR indicators are as follows (Weber, 2010). Drivers (agriculture, extractive industries, chemistry, transportation, urban development, trade, consumption patterns, tourism, demography); Pressure (over-harvesting, flows to the environment, land use, waste); State (description of the environment, structure and functions of ecosystems, degradation - improvement); Impact (change in the environment, effects on economy, habitat, population); Response (nature protection, control, ecosystem management, pollution abatement, land planning, fiscal policies, payments for ecosystem services, or PES). A first attempt to clarify the respective coverage of the DPSIR model by ENCA, Planetary Boundaries, MAES, SEEA, Ecological Footprint, and HANPP is shown in **Figure 1.2** (JL Weber, personal data; details in **Supplementary Box 1.1**). The diagram indicates that so far pressure (and driver) indicators have preferentially been employed.

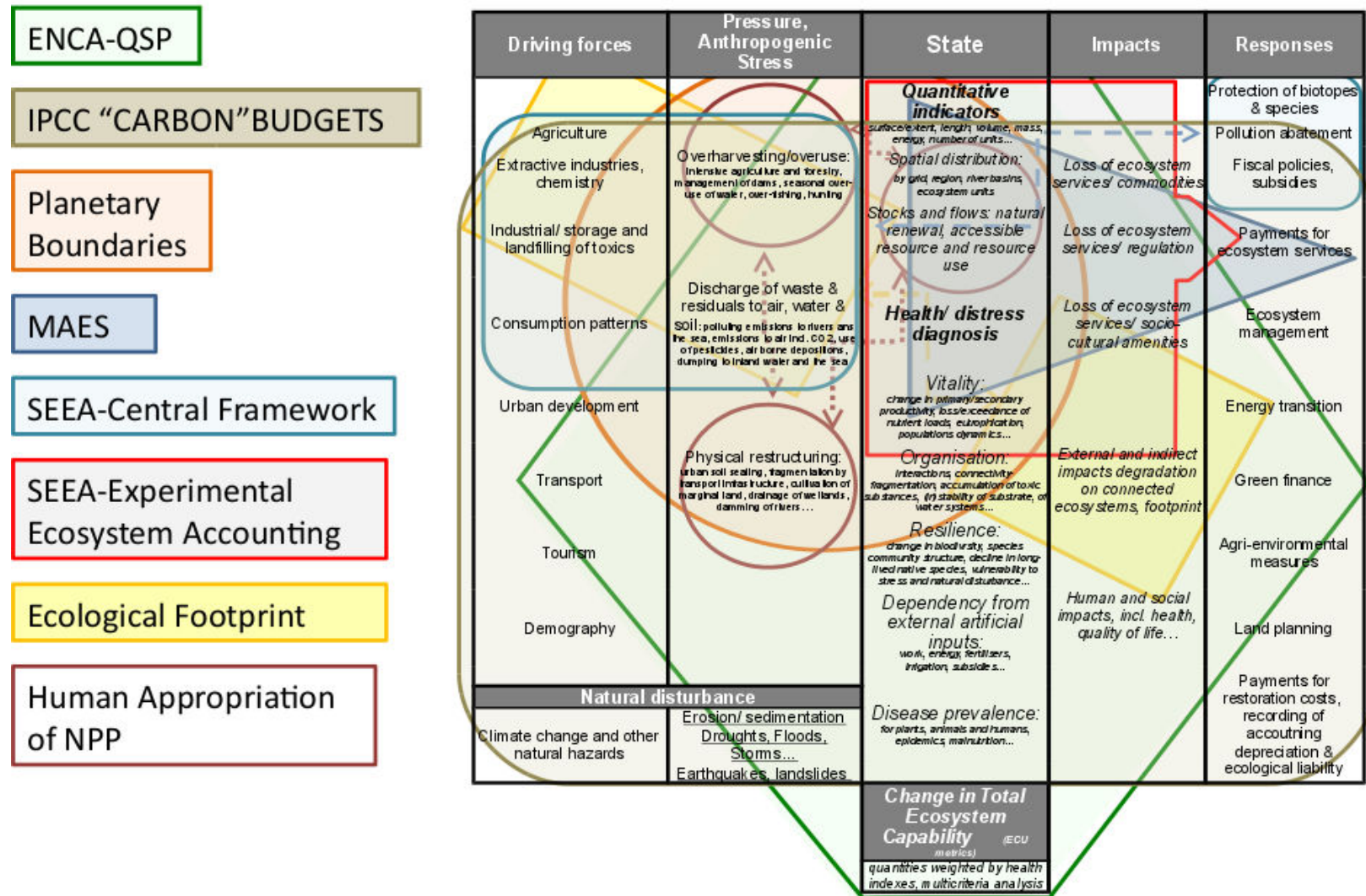


Figure 1.2. DPSIR model with main focus on ecosystem assessment and accounting frameworks. DPSIR components are differently covered in environmental approaches and programs.

(2) Define what type of baseline (benchmark) is used in each of the analysed methodologies, namely,

- the current state (*status quo*) to measure the extent and rate of change of the ecological state (degradation, maintenance, improvement);
- evolution with respect to a previous state (e.g. CO₂ levels in 1990; preindustrial era for planetary boundaries).

1.2.3 Discussion

The Discussion focuses on the following aspects: considering main trends in environmental evaluations, identifying limitations and problems in current methodologies and policies, and arguing on the necessity to optimizing environmental evaluation tools so as to target intrinsic value of ecosystem capital in the near future.

1. Trends in environmental evaluations

The analysed items cover a broad range of complementary and sometimes contradictory approaches to environmental assessment. A majority are use-value oriented, with the more recent emergence of cost evaluation methodologies. Ecosystem services is a case in point, the aim being achieving service aggregation in monetary units while avoiding double counting or no counting of services due to their interconnected and multifunctional nature (Weber, 2018).

Concerning the DPSIR system, the state of environmental reporting is traditionally structured by the Pressure-State-Response (PSR) framework proposed by OECD in the 1980s. In the 1990s, PSR has been refined by the European Environment Agency into DPSIR for Driving Forces-Pressure-State-Impacts-Response framework. Both underlie the various environmental information systems, would it be explicit or implicit. Beyond formal differences of limited importance, PSR and DPSIR can be seen as either a convenient way of presenting environmental information to policy makers (OECD) or a more or less deterministic sequential model where drivers (economy) generate pressures (pollutions, resource depletion) which are the cause of degradation of the state (natural resource, nature condition or health) resulting in impacts (loss of services, impacts on connected systems) which require policy responses. Such responses can address either causes or effects. They are typically norms or taxes on drivers in order to reduce pressures, hence improving state and mitigating impacts. State can be protected by nature conservation schemes and impacts offset by e.g. financial measures. Furthermore the DPSIR analysis, designed to assess the progress of environmental policies, has evolved from (1) measuring pressures separately with the inherent

difficulty to aggregate them in a composite pressure index (PSR system; SEEA, 1993) to (2) adopting an ecosystem health approach centered on state variables that more explicitly connect the state of the ecosystem with the consequences of human activities and their cumulative impacts (Impacts and deduced Pressures).

More generally, our analysis shows that there is

(1) A better recognition of ecosystem service delivery with degradation considered as “loss” of service with aggregation through monetary valuation (market or shadow prices), i.e. a narrow capital approach (World Bank 2011, UNEP 2014, VANTAGE; see also Ten Brink, 2011; Schultz et al, 2015; Guerry et al, 2015).

(2) An integration of ecosystem capital to the conventional capital approach aiming at the maintenance of the ecosystem potential to deliver services, the degradation being considered as “loss” of its overall performance at delivering services and resilience. The aggregate relates to the ecosystem itself, valuation considering primarily restoration costs of ecosystem functions (Weber, 2018) and should *in fine* inform the intrinsic value of the ecosystem capital.

2. Inconsistencies and incongruities

The main issues are presented below.

(1) There are major gaps and heterogeneity in the generation of data (collection, access, processing - and in particular scarcity of time series and / or spatially generated data), and inconsistencies in the quality of the statistics on which measurement of the physical state of ecosystems (degradation or improvement) rely on. Despite a sort of metric obsession facilitated by the continuously expanding satellite and geospatial data, and the structuring efforts at

- international level (see the Global working group on big data and its guidelines “Earth Observations for Official Statistics Satellite Imagery and Geospatial Data Task Team report 2017 with pilot projects hosted by the UN Global platform for Official Statistics, or the ESSnet Big Data 2 – Earth Observations sourced by Eurostat, or the LandSat and Copernicus programs of ESA) and
- national level (the IGN platform to coordinate and integrate primary producers of georeferenced data in France, to name just one case), such developments are in the exploratory stage of assessing the potential of remote sensing data and are working at elaborating a methodological frame and reference system (INSEE, 2019). In parallel, difficulties persist in generating and accessing other type of data that are statistically fine mapped, namely water and river, biodiversity, agriculture, and socio-economic variables.

On the other hand, private sector strategies to data information for decision-making in incorporating the natural capital assessment in corporate financial, risk, and environmental management is progressing. The objective is to understand and manage the impacts and dependence on natural capital (Natural Capital Coalition, 2019). In their hands, the data barriers that limit the use of data are less an absence of data and more an absence of flow between data providers, data collators, and data users.

(2) Biodiversity evaluation raises concern because first, there is no consensus definition in the analysed lot of items (**Supplementary Table 1.1**) despite the fact that the standard definition has been set by the Convention on Biological Diversity (2003), and second, evaluations are produced while there is neither consistent national or regional biodiversity monitoring nor a harmonized observation system based on exhaustive, systematic, and regular data on biodiversity state *stricto sensu* (genetic, population / demographic, or phenological traits or trends; Brooks et al, 2002; Dornellas et al, 2014). For this reason, such variables, and in particular extinction rates (presently the global extinction risk was estimated at 2.8 percent; Urban, 2015) or Percentage Disappeared Species should be avoided. Rather, the literature suggests that certain Essential Biodiversity Variables (EBV; Pereira et al, 2013) best answer why and how to measure biodiversity. Preference should go to variables that capture major trends in biodiversity change through alteration in ecosystem structure (remote sensing of habitat structure, fragmentation levels, frequency and aggregation of different ecosystems) and their functioning (NPP, nutrient retention, disruption regimes etc). Habitat loss and degradation and land use change are the primary cause of substantial changes in species abundance, distribution, and interaction (Dirzo et al, 2014) and can serve as a reasonable proxy indicator of biodiversity states (see also Brooks et al, 2002 for correlations between native vegetation extent and biodiversity scoring). Additional correlations could be worth investigating based on the carrying capacity of human populations at given territorial scales (see Ecological Footprint approach).

(3) While more methodologies tend to integrate the DPSIR model, the reference for environmental indicators and statistics, in assessing the ecosystem health by measuring the current state of resources and the environment, identifying pressures, and monitoring change (impacts) over time, there is an obvious need to focus the political and societal demand on strong sustainability in order to define a general and ambitious goal for environmental evaluation.

3. Optimizing environmental evaluation tools (targeting intrinsic value of ecosystem capital)

Chapter 1 State-of-the-art in environmental evaluation since 1990

A general and ambitious goal in environmental evaluation for aid to decision-making should allow upgrading the existing tools. This implies provisioning generic data processing platforms that collect and update data sets to enable systematic comparisons of ecosystem capital state in space and time, and focusing the political and societal demand on achieving no net ecosystem degradation. In other words, the goal is targeting the intrinsic value of the ecosystem capital (that can not be replaced by economic values) rather than the use value. The only environmental evaluation approach that target the intrinsic value of the ecosystems is ENCA.

A comprehensive protocol for aid to decision-making (administrative, economic, political) would consider achieving a thorough description of physical and digital resources for a given territory, capable of monitoring the quality of management and internalizing constraints by all actors in a variety of economic, political, and institutional contexts.

Upgrading the existing methodologies must consider systemic accounting approaches that ensure the maintenance of ecosystem capacities (no net ecosystem degradation) as the bottom line. By estimating the ecosystem potential (or capability) one can engage with strong sustainability policies. Establishment of an ecological balance sheet in physical units and aggregation of accounts in intrinsic ecological value units are essential for both comparison over time and integration in national accounts.

A systemic and exhaustive accounting for the ecosystem capital established on a robust statistical base would include accessible resources description, production, and supply, the intensity of their use (consumption, accumulation, and trade), with corresponding remunerations, profits, taxes and subsidies, financial flows and assets, and debts originating from the entire process. This could subsequently be translated into financial costs of depreciation and conservation / compensation protocols, allowing calculating the full costs of products and restoration costs.

This implies a triple constraint in implementing data policies specifically dedicated and adapted to ecosystem capital assessment:

- Evaluate ecosystem degradation through norms that work as social signals, are not prescriptive, and mobilize financial levers (rather than norms external to society);
- Use of homogenized statistical norms (statistical socio-ecological standards and units), allowing the integration of physical unit data in macro-economic models (such as those on C quotas) and
- Setting up accounting norms (accounting standards with units of measurement to quantify in relative terms the degradations / improvements) of the physical state of ecosystems at various scales

(based on data that are exhaustive, regular, coherent, transparent, and verifiable) enabling local-to-global comparisons to implement systematic ecosystem capital amortization.

The present analysis allows sketching an optimized tool for the evaluation of the intrinsic ecological value as an aid to decision-making (administrative, economic, political) that is expected to be inclusive, integrated, and actionable simultaneously in distinct socio-economic contexts in order to :

- Organize information at the scale of functional ecological units (water basins, mountain ranges, ecosystem types, etc.) according to core accounting grids;
- Provide methodology for integrating ecosystem degradation into ecosystem capital accounting standards that describe and measure stressor effects with comprehensive and contextualized indicators that out weight juxtaposed aggregated indicators (in the form of lists or dashboards);
- Value with multi-scalar environmental diagnostic tools the ecological capital in physical units (gains / losses) as a measure of improvement / degradation;
- Estimate depreciation costs (polluter-payer and other unpaid costs) and the programming of ecosystem capital amortizing costs (natural resource degradation), as well as investment and financial depreciation costs (ecological restoration or avoidance costs);
- Quantify in synthetic manner the externalities of the economy and internalize them in decision processes through annual reporting.

That expertise will become invaluable in

- Detecting early warning signals of ecosystem capital degradation;
- Assessing the impact and effectiveness of public policies, planning decisions and management, and economic activities on the ecological potential;
- Addressing issues such as estimation of full costs (of use, production) and anticipation of financial risks for public institutions (tax base / liability, keeping ecological balance sheets as a market condition) and private companies (financial rating, investment risks, etc.);
- Serving as basis for upgrading legal and institutional frames aiming at the full amortization of the ecosystem capital.

To turn this intellectual exercise into an actionable research object, I worked out, together with the ENCA project team, the organization of a workshop on upgrading the environmental evaluation process and tools. The proposal is detailed in the **Supplementary File 1.1**. I anticipate that the recommendations of the workshop panels might support some of the conclusions to which I came in this document.

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Chapter 1 State-of-the-art in environmental evaluation since 1990

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1.2.5 Supplementary documents

Link to supplementary Table 1.1

https://drive.google.com/open?id=1yWuRStKnlqS7XtDp2n_dE1v_DpSI7A3

Supplementary Table 1.2. Core indicators targetting ecosystem to natural capital relationship.

Glossary	Definition	Measurements	References
Biodiversity	"the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems."	<ul style="list-style-type: none"> -Diversity richness -Endemism -Habitat alteration and land conversion from natural state -Threatened or extinct species as a share of total species known -Area of key ecosystems -Protected areas as % of national territory and by type of ecosystem -Protected species -Extinction rate -Biodiversity intactness 	TEEB 2008 MA, 2000 OECD, 2001 Steffen et al., 2015 Rapport et al., 1998 Müller and Burkhard, 2010 Pereira et al., 2013
Ecosystem capability	The overall potential to deliver any service in a sustainable way without reducing the potential for other services.	<ul style="list-style-type: none"> -ECU: accessible resources with index of sustainable use and ecosystem health -Ecosystem condition -Ecosystem changes: degradation or improvement 	Weber, 2014 Ouyang, 2016
Ecosystem services	Are the benefits people obtain from ecosystems.	<ul style="list-style-type: none"> -Supporting services -Provisioning services -Regulating services -Cultural services 	TEEB, 2008
Matter balance	An ecosystem property, including the imports, exports, storages, and the internal flows of nutrients.	<ul style="list-style-type: none"> -Nutrient loss: N -Storage capacity: N:C balance 	Müller and Burkhard, 2010 Weber, 2014 Rapport et al., 1998 Pereira et al., 2013
Water balance	An ecosystem property, including the imports, exports, storages, and the internal flows of water.	<ul style="list-style-type: none"> -Water flows: transpiration per evapotranspiration 	Müller and Burkhard, 2010
Energy balance	An ecosystem property, including the imports, exports, storages, and the internal flows of energy.	<ul style="list-style-type: none"> -GPP -NPP -Entropy -Output by evapotranspiration and respiration -NPP/soil respiration -Respiration per biomass 	Müller and Burkhard, 2010 Rapport et al., 1998
Intensity of use	An indicator assessing the impact of sectorial economic activity on the resource.	<ul style="list-style-type: none"> -land -water resources (abstractions/available resources) -forest and agricultural resources (actual harvest/productive capacity) -livestock and fish resources -energy -sustainability (resource use / productivity) 	OECD, 2001 Müller and Burkhard, 2010 Weber, 2014 Verburg et al., 2015 Karl-Heinz et al., 2013
Natural capital (NC)	Economic metaphor for the limited stocks of physical and biological resources found on Earth. Commonly NC is used to refer to all types of environmental assets. Accordingly, NC includes mineral and energy resources.	<ul style="list-style-type: none"> -land stocks -water stocks -agriculture -minerals -energy -biological 	SFEA, 1993 Weber, 2014
-Ecosystem assets -Ecosystem capital -Environmental assets		<ul style="list-style-type: none"> -Non monetary -Monetary 	

Supplementary Box 1.1

Confronting various information frameworks against DPSIR without being exhaustive and without adopting the sequential standpoint which focuses in practice on Driver and Pressure (see also Table 1.1 and Chapter 2 section 2.1 for more details and references).

1. The Ecological Footprint (EF) is a variant of carbon accounting where “biocapacities” needed for resource supply and CO₂ sequestration are converted into hectares. It is based on D and P (only GHGs). S is not addressed (no pollution, no biodiversity...). I and R are presented in very theoretical terms: additional Earth surface needed and (similarly to IPCC) reduction of resource use.

2. Planetary Boundaries (PB) address P and S. However, individual indicators cannot all be clearly defined in that way. For example, biosphere integrity seems to be a state indicator, but its metrics is based on pressure (by land use, by nitrogen depositions, biodiversity erosion) on biomes. Freshwater use and land use are at this stage more P or even D indicators than S. In terms of policy target, PB’s main focus seems to be controlling P. This is illustrated in the translation of PBs in “Critical Pressures” and then in “Planetary Quotas” (Meyer and Newman, 2018) where the levy is mostly the abatement of P to respect the boundary limits.

3. MAES (Mapping and Assessment of Ecosystems and their Services) and VES (Valuation of Ecosystem Services, like in WAVES, NCA&VES...) are models based on ecosystem state accounts and maps (S) to assess ecosystem services (provisioning, regulating, and socio-cultural) and their condition (I) with the purpose of valuing them in money as a way of emphasising their importance (R).

4. The SEEA Central Framework is clearly focusing on D, P, and R which are structured in close relation to National Accounts and in particular the UN Central Products Classification (CPC). Natural assets are recorded as reservoirs of resources for the economy. There is no carbon account in the SEEA CF, but accounts of products composed of carbon (e.g. timber). Water is considered as a commodity which can be abstracted for use from assets. On P, the SEEA CF maps emissions against ISIC, the International Classification of Industries, with the purpose of broadening the scope of data for macro-economic modelling. Concerning R, environmental protection expenditure accounts are well developed.

5. The SEEA Experimental Ecosystem Accounts covers ecosystem state which is described as “extent”, “condition”, and provider of ecosystem services. The latter plays in fact a central role as “ecosystem capacity”, defined as the capacity of delivering a bundle of services over time. Ecosystem services are valued (R) and ecosystem total wealth in money is calculated according to the standard capital model of the economic theory where capital value equals the net present value of expected profits over a conventional period of time.

6. The ENCA-QSP framework covers directly and indirectly DPSIR. Oppositely to the usual approach of the DPSIR sequence, ENCA starts explicitly from S, described in terms of quantitative and qualitative accounts based on diagnosis inspired by the ecosystem health theory of D. J. Rapport (Rapport et al, 2009).

The aggregated indicator, the Total Ecosystem Capability, is defined in terms of ecosystems functions’ robustness and resilience. Uses of ecosystem services embedded into production and returns are recorded according to socio-economic statistics and are therefore closely connected to D and P. Considering I, beyond ecosystem services, indirect impacts of one given ecosystem on other ecosystems functionally connected, neighbouring or embedding are important characteristics. ENCA development will allow integration of ecological balance sheets to record governments’ and economic actors’ liability to ecosystem degradation or improvement and ultimately ecosystem depreciation in financial accounting standards.

Other information frameworks (not directly addressed by the comparative analysis)

7. HANPP, the Human Appropriation of Net Primary Productivity (the biomass) is based on D and P (harvests, land restructuring...) for its variables while defining a reference value in terms of state, S being the potential natural productivity in the absence of human intervention. HANPP assumes a correlation between non-appropriated biomass and biodiversity.

8. MFA: Material Flows Analysis or Accounting. Developed in particular in industrial countries for the purpose of measuring “resource efficiency”. Key elements in the OECD Green Growth model or in the European Union resource efficiency strategy. Their focus is on D and P. No attention to S and I.

9. IPCC reporting. It is a variant of MFA focused on “carbon” or “CO₂ equivalents”. Contrarily to MFAs, the central point is S (the global warming target expressed as “+ 1.5 degree”...) and R is the purpose of the framework (countries commitments to reduce their GHGs emissions). In recent years, work has been undertaken on I, the impacts of global warming (storms, floods, droughts...).

Supplementary File 1.1

Outline of an international workshop on environmental evaluation 2020

Environmental impact assessment methodologies – societal and political demand, and state-of-the-art debate

Abstract

How to measure the degradation of nature to determine the depreciation / amortization of its use, not recorded in the balance sheets of the nations? We have developed a proof-of-concept tool of "ecosystem accounting" to evaluate the state of the ecosystem capital. Our systemic approach is one among a competing set of tools for environmental evaluation. There is currently no contradictory academic debate about the ability of these tools to inform and guide political and economic decision-making. There is no established peer-reviewed expertise in environmental evaluation.

This project challenges the situation. It consists of (1) organizing an international workshop as a first confrontation between experts in the field (science-to-science) and concerned decision-makers and users (science-to-policy) and (2) providing a workshop preparatory research to position IXXI in the strategic landscape of environmental evaluation. The project will enforce the ongoing cooperation with STEEP in Grenoble (territorial resilience, modelling) and EPFL in Lausanne (Rhône watershed, data policies, systemic risk assessment).

Rationale

Most countries are not capable of adapting their resource use to the basic needs of their populations and maintaining life-supporting natural systems on which societies depend on (Sas, 2015; DGS 2018). They do not incorporate ecosystem capital degradation in accounting standards. The overall degradation of natural systems resulting from economic activities is therefore equivalent to unpaid costs corresponding to ecological debts (Bartelmus 2010; Dasgupta 2010; Vanoli 2005; Arguello et al, 2018; Weber 2014)). These debts are virtual as long as they are not measured and recorded in ecological balance-sheets in order to be offset. The consequence is the accumulation of ecological debts over time, which creates economic and political risks of which institutions as well as business are more and more aware of (E-Risk 2012; Steffen et al, 2015; Weber 2018). A range of approaches, methodologies, and instruments of environmental evaluation has emerged in the last 20 years (Mazza et al, 2013; Weber 2018) present the pros and cons of these methodologies in general terms). They tend to integrate the ecological capital into economical accounting frameworks, national accounts firstly and now financial accounts (Caldecott 2013). However, it is difficult to understand who is who in the field : the underlying goals of these environmental assessments are manifold, covering anything from integrated resource management (green growth) to exhaustively describing the state and ecological value of ecosystems (strong sustainability). The corresponding tools range from communication indicators to more operational metrics (Arguello et al, 2018). Their capacity to provide aid to decision-making needs science-based validation. There was so far no in depth comparative analysis of these methodologies, which makes hazardous attempts to introduce them in decision-making. Rather, there is grey competition and lobbying between international institutions / agencies to promote such tools. In addition, most of them have not been evaluated through the peer review process and funding resources are being used inefficiently. In brief, it is time mobilizing the scientific community on the issue.

Several of the above issues have been addressed by the contributors to the project:

- the creation of a Natural Capital Network initiated in 2015, whose members are organizers, members of the scientific committee of the workshop, and workshop contributors;
- within the network or in collaboration with other IXXI partners, the following seminars have been organized to tackle problems such as *Territorial complexity: legal and data challenges* (2018); *From data ecosystems to ecosystems of data* (Lausanne, 2016); *Regards croisés sur la santé commune. Du Contrat naturel aux savoirs traditionnels des Indiens Kogis (et retour)* (2018);

Chapter 1 State-of-the-art in environmental evaluation since 1990

- IXXI members have worked on aspects concerning big data, the production of the ecosystem accounting methodology (PhD thesis), and production of an exhaustive inventory of indicators on public health, social cohesion, and state of ecosystems. Two research projects on ecosystem capital accounting in the AURA region are in the evaluation stage;
- cooperation with STEEP, INRIA Grenoble (in particular their expertise on analytical tools to evaluate the complete footprint of a territory in its interactions with the rest of the world) and with EPFL/LASIG (on the Swiss part of the watershed);
- the project feeds the new CNIG / IGN geo-platform program for sovereign geographical information at the regional level allowing a description of the territory for all strategic resources (physical, digital, institutional) and the design of new services that create added value by promoting interoperability around major equipment projects on the one hand, and by identifying the vulnerabilities as well as the assets of target territories and promoting a greater accountability of the decision-makers and actors, on the other hand.
- the PIA project Biovallée (Drôme) aims at evaluating the ecological capital at the « communauté de communes » level and implement crowdsourcing.

In summary, IXXI has acquired appropriate expertise internally and through collaboration within ENS (EVS, CBP), and regionally (STEEP Grenoble and EPFL; and some primary data providers agencies). What precedes assembles reasons, arguments, and achievements that explain why we consider timely and benefiting to IXXI the organization of the proposed workshop and its preparatory strategy in order to ensure a science-based controversy between the promoters of the main methodologies in the field and data providers, decision-makers and users.

The workshop

The ambition of the workshop is to address some of the identified bottlenecks by bringing together scholars, decision-makers and a range of public service users, international -to-local agencies and institutions, and developers of environmental evaluation methodologies and tools. The following approach has been considered.

1. Identify the problem - wide use of incomplete and / or misleading indicators of production, income, consumption, saving, investment, and debts that ignore externalities generated to the natural capital.
2. Define an objective – target “no net ecosystem degradation” and ultimately the amortization of the natural capital (“strong sustainability” defined in terms of “critical natural capital”).
3. Compare and propose actionable methods / tools fulfilling user expectations, needs, and the target objective by evaluating the following methodological categories: reference value or boundary approach; ecosystem services approach; system accounting approach.
4. Identify resources and cooperation capacity to upgrade best-fit tools and protocols that inform and sustain policies (such as data policies) at various scales. Inform on ongoing activities in the private sector.

The workshop sessions are:

S1 - Political, administrative, societal, economical demand - the a priori vision of decision-makers and users, and current indicator systems.

S2 - Pressure based indicators : limits of use, footprints - ex. Environmental Norms, Excise taxes, Planetary boundaries, etc.

S3 - Ecosystem services assessment and valuation - methods, examples, interest and limits of monetary valuation.

S4 - Ecosystem maintenance (systemic) approaches - methods, examples, ecological valuation and ecosystem degradation costs, Ecosystem Natural Capital Accounting presentation (Rhône watershed)

S5 - Ecological debts and systemic risks . Operational responses (expected and provided) by ecosystem assessments including :

- standards, taxes, public investment, depreciation of natural capital, full cost of products, financing of restoration and conservation, payment for ecosystem services, ecological and financial risks, green finance, conditionality of public contracts, integration of international indicators / multiple scales.

S6 – Data Resources - IT, platforms, data and Big Data.

Of note, environmental diagnostics are performed on platforms managed by private operators specialized in the production of geospatialized information through processing of massive data on stocks and fluxes of accessible resources in quasi-real time primarily for profit-making (Google Earth Engine and, more recently, the European network “Earth Intelligence and Planetary health intelligence”).

The expected results of the workshop are the identification of strengths and limitations of current methodologies, providing recommendations and guidelines for upgrading them, make public the work document in a peer reviewed journal, and organize partnerships between decision-makers and scholars to implement the recommendations through research project design. We anticipate that the IXXI developed methodology will serve as reference for users that target no net ecosystem capital degradation (also see section on Relation to complex systems).

Pre-conference work

Feeding the workshop debate. Seen the multi-actor nature of the workshop and the two day program, a preparatory and co-construction activity is necessary. An inventory of environmental assessment methodologies (3 major classes, 25 items; see sections S2-S3-S4 above) is underway aiming at understanding in context their respective advantages and limitations. On that basis, future work will consist in elaborating a state-of-the-art report including the synthesis of indicators on public health, social cohesion, and the environment (according to the results of the workshop "Complexité territoriale, enjeux juridiques et numériques" 2018).

Report available ahead of the workshop (January 2020), to be completed with recommendations of the Workshop, followed by publication in a scientific journal (co-authored by the promoters of the workshop and moderators of S1-S6 sections).

Valorization of the proof-of-concept tool. The automation of calculations for ecosystem accounts (statistical and geographical data) is necessary to simplify the treatment chains by integrating them into a single environment (e.g. jupyter notebook) to facilitate their exploitation. The process consists of integrating the modules of the various geomatics and spatial analysis software packages (eg GDAL, QGIS SAGA-GIS). The automation is an important element in demonstrating its technical capabilities and interest to the users (such as Agence de l'eau, DRAAF, CNR, MRC etc). On the societal side, the tool can be used to perform pedagogic simulations putting the Rhône basin in a situation of overexploitation of strategic resources and under various levels of carrying capacity.

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Chapter 1 State-of-the-art in environmental evaluation since 1990

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1.3 Planetary boundaries revisited, article in “Lancet Planetary Health”

The comparative analysis I made in the previous section indicates that environmental evaluations target and share most of the time three strategic resources: land, water, and biomass. This is not surprising, because these are basic physical constituents of ecosystems.

Seen the fact that Planetary Boundaries (PB) concepts is the new-comer in the field and is gaining strong scientific and political weight/attention, I wanted to understand how the three basic components, (land, water, and biomass) are considered and how they connect with the other components of the PB system. This seemed necessary, the PB system having so far been described as an assemblage of rather interdependent elements.

Upon analysis, we conclude that PB can be trimmed down to two main groups of parameters. This makes it easier to understand what is at stake in the first place and what difficulties could limit an actionable implementation of the PB system (or part of it).

The published Comment is inserted below, and is illustrated with a diagram (**Figure 1.3**) showing the proposed two-component aggregation of boundaries, namely agriculture and global pollution, and a panel (**Figure 1.4**) analysing the cocktail effect. The figures were part of the submitted manuscript that had to be modified to fulfil editorial rules (the original text is presented in Supplementary File 1.2).

Agriculture and global physicochemical deregulation: planetary boundaries that challenge planetary health



Clear identification and characterisation of social and ecological challenges is crucial to define priorities and allocate limited resources,¹ and to work out solutions and tools for their implementation. Research should serve to increase human understanding of those challenges and the objective interdependencies among them. The resource concept constitutes a unifying theme across disciplines and sectors, such as the Sustainable Development Goals (SDGs),² a first point of reference for systemic change. Therefore, the resource approach becomes the most effective tool to address the surrounding issues of planetary health.² The notion of social and planetary boundaries^{3,4} is a complementary conceptual framework that defines safe operating boundaries for its nine dimensions (climate change, ocean acidification, chemical pollution, nitrogen and phosphorus loading, freshwater withdrawals, land conversion, biodiversity loss, air pollution, and ozone layer depletion). The framework's goal is to help achieve a sustainable future for humanity³ by suggesting safe limits to the nine dimensions. This framework allows to measure such limits by using a combination of warning sign, threshold, and regime shift indicators. The transgression of the framework's boundaries through the misuse and misallocation of resources has undesired consequences to global functions and cycles of Earth's life-supporting systems and is leading to the disturbance of complex ecological and physicochemical equilibria to a degree beyond their natural resilience. The framework and its use has sparked interest within the scientific, societal, and political communities.³⁻⁵ By showing the interconnectedness of the boundaries, it is possible to clarify planetary health issues and give them meaning in political and societal terms. The framework can be condensed down to a two-component system: agriculture and physicochemical deregulation.

Agriculture is both the greatest labour provider and resource consumer (land-use change and biomass appropriation). Agriculture and associated food systems (from fork to farm)⁶ are the biggest contributors to the stressors of planetary boundaries, including changes in land and water systems, ecosystem alterations, fertiliser chemical flows, air and water pollution, and climate change. More

specifically, agriculture entangles the key biogeophysical building blocks of the biosphere—biomass, soil, and water—which need protection from environmental and management related risks through policy decision making. This is important because the geopolitics of biomass, water, and land are driving the new resource scarcities,^{7,8} which are causing conflict between national security and international order mechanisms.

Global physicochemical deregulation is a pervasive and hidden global challenge. It is generated through ocean acidification; more general atmospheric, land, and water pollutions; waste accumulation; and climate change. The corresponding chemical intensification affects societies daily on a large scale: 144 000 types of chemicals are manufactured, with a 500-fold increase in volume over the last several decades.⁹ The result is a so-called mega-pollution system consisting of complex chemical cocktails in the environment (eg, pesticides, endocrine disruptors, antibiotics, and microplastics) that interact with the natural chemical world¹ and change over space and time. This makes it hard, and sometimes impossible, to measure, understand, and control the exposure to a mixture of chemicals for which safe limits for polluting factors have to be estimated with regard to exposure to any single chemical.¹ Current standards are based on studies that have been done on one chemical at a time. Mixing different chemicals can result in additive, synergistic, or antagonistic effects. For example, simple chemical mixtures that consist of five pesticides found in food commodities showed additive to synergistic genotoxic effects in cultured cells.¹⁰

Consequently, a detailed knowledge of the response to all possible combinations of chemical risk factors implies dividing the limit value for an individual factor by the square root of the number of factors under consideration. Although this calculation would be unrealistic in practice, it suggests that the adoption of limit values two or three orders of magnitude smaller than those estimated by single factor impact studies¹ is relevant to legislation. However, this is a problem: prevention protocols, standards, and legislation are not well adapted to the scale of risks operating in constantly evolving environments.

Comment

Agriculture and physicochemical deregulation act in synergy, are first order systemic risks to planetary health² with long-term effects on the society. For example, the health and economic burden of conventional food systems and global pollution is estimated to cost trillions of US dollars annually and contribute to millions of deaths globally.^{11,12} Low-income countries are the most affected. We argue that the health and economic burden constitutes a systemic challenge (so-called health bubble), which government and society will have to address over the next decade.

Therefore, we call for, first, coordinated measures that target planetary health through coherent chemical simplification (effectively reduce, redesign, and recycle to make better use of a smaller number of different chemicals) in transitions to societies that decrease their use of resources and are resource efficient.^{2,13} This measure could also help to slow down climate change. Second, we call for the integration of sustainable agriculture practices into food system transitions⁹ with priority on land tenure and value and the food-health nexus.

These measures are at the apex of further changes in joint environmental and societal policies that articulate physical and social boundaries⁴ in a resource stewardship approach. They are guided by simple, universal, and indivisible principles in support of a civilisation contract.² The goal is to measure the total costs to society of no action versus partial-to-complete phase-out of large sets of chemicals and unsustainable practices. This phase out would help to align equity in health and wealth within the limits of the biosphere. By addressing the health concerns associated with agriculture and physicochemical deregulation it could be possible to devise laws that enable fair access to and

allocation of resources, a process for which WHO¹³ and the International Resource Panel³ could ensure the initial coordination role.

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We declare no competing interests.

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«Health bubble »

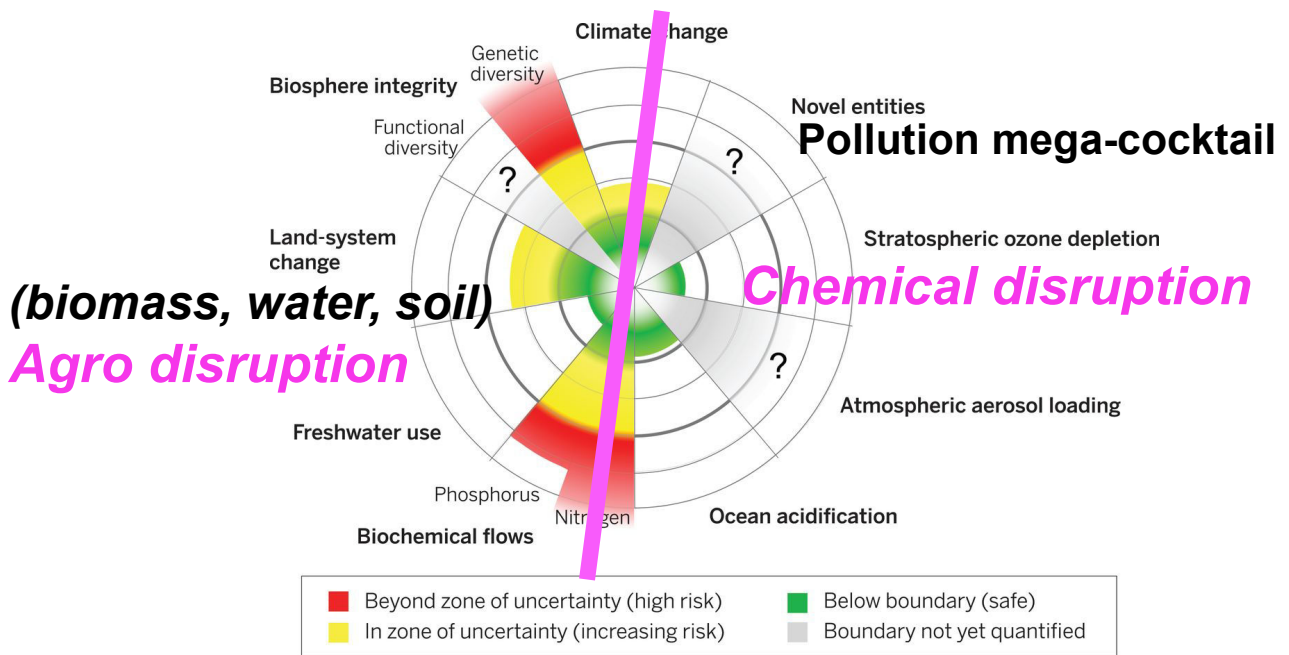


Figure 1.3. Planetary Boundaries revisited (adapted from Steffen et al, 2015). The analysis of individual boundaries allows establishing logical interdependences and their assignment to two main categories, namely agriculture and global pollution.

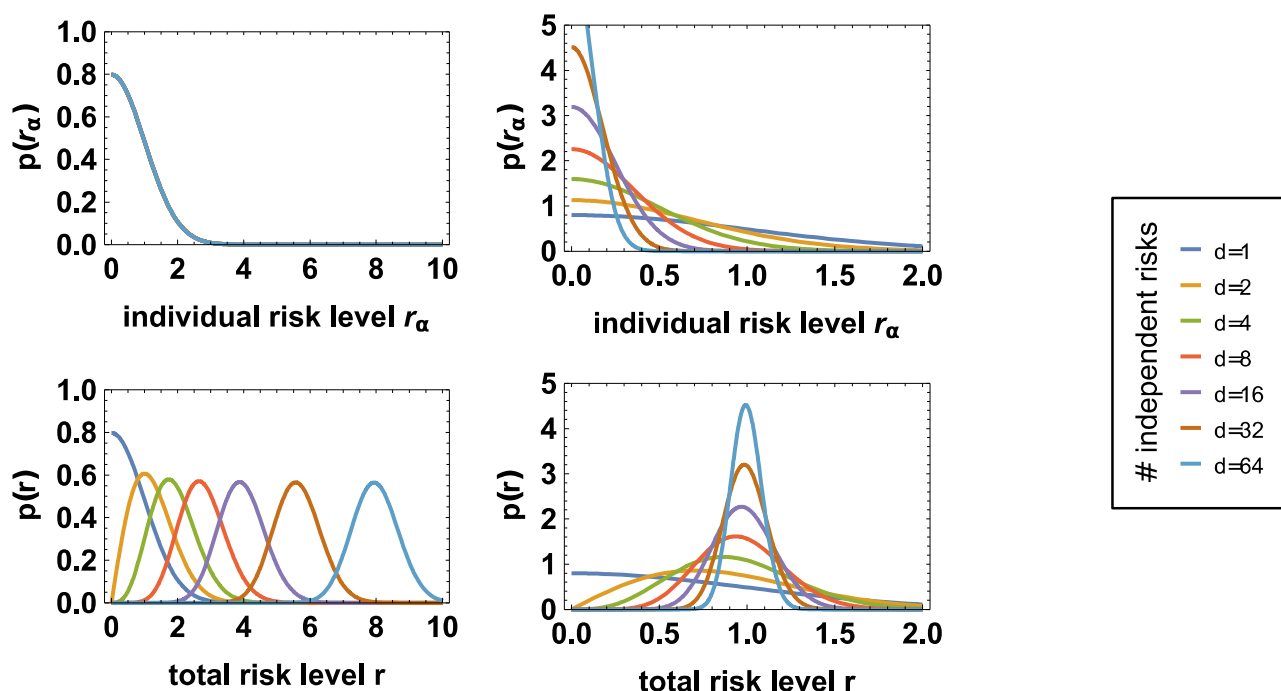


Figure 1.4. Safe operating limits in high-dimensional risk spaces by representation of multi-dimensional threat levels (courtesy of Ralf Everaers).

Establishing thresholds (limit values) through risk analysis of health harmful effects consists in applying a typical safety factor 100 (Lomborg, 2001). Consider d independent stresses, each assessed on a scale where 1 represents a safe boundary in the case where a given risk factor is taken in isolation. Examples could be radiation doses or concentrations of various toxic substances to which organisms or ecosystems are exposed. In this case, risk exposure is characterized by a d -dimensional risk vector, $\vec{r} = \{r_1, r_2, \dots, r_d\}$ enumerating the individual stress levels. The norm $|\vec{r}| = \sqrt{\sum_{\alpha} r_{\alpha}^2}$ is a naive, but plausible, estimate for the total stress level in a complex system with a large number of components exposed to independent stresses: exposure to d individually tolerable stresses $r_{\alpha}=1$ increases the total risk level to \sqrt{d} . Now assume that all risk factors r_{α} are managed independently such that their values follow a normal distribution of a width $[\Sigma]$, which is to be chosen for legislation (top row of panels). In this case, the Maxwell-Boltzmann distributions of the estimated total stress or risk level (bottom row of panels) are straightforward to calculate as a function of the number, d , of independent stresses. The panels in the columns consider two different scenarios.

On the left, we illustrate how an intolerable total risk level of the order of $\sqrt{d}=1000$ arises from the controlled release of $d=10^6$ substances at levels which can, with $\sigma=1$, be safely supported individually.

On the right, we show that in order to avoid the accumulation of risk due to a large number of different stresses, individual risk levels need to be managed on levels which are a factor of $\sqrt{d}=1000$ smaller than evaluated by single factor impact studies.

Supplementary File 1.2

Intersecting planetary health and planetary boundaries reveals the double challenge of agriculture and global physico-chemical deregulation

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Clearly identifying and characterizing challenges is crucial in defining priorities and allocating limiting resources¹ to work out solutions and tools for their implementation. Research must serve to increase human understanding of those challenges and the knowledge of objective interdependencies among them.

The resource concept constitutes a unifying theme across disciplines and sectors, such as the SDGs², a first order “leverage point” for systemic change³. It therefore can become the most parsimonious instrument to address the issue of planetary health² and as a consequence reframe social rights issues and public health policies.

The planetary boundaries concept is a complementary lever defined as a safe operating space for a sustainable humanity⁴ by setting limits to a series of biological and geo-physical variables that contribute to global functions and cycles of the Earth system and restrain current rush on and unfair access to resources.

The planetary boundary framework with its warning signs, thresholds, and regime shifts measures values of biosphere integrity, global pollution, and climate change. Transgressing such boundaries through the misuse and misallocation of resources is leading to perturbation of complex ecological and physico-chemical equilibria to a degree beyond their natural resilience⁵. The geo-physical and chemical boundaries can be rather accurately quantified (typically, air emissions such as CO₂ concentrations, ocean acidification, material and main chemical flows, energy stocks and flows, and water use, while the ecosystem components (i.e. the genetic diversity and ecosystem functional diversity, ecosystem stocks and changes in stocks, ecosystem services, etc) are more recalcitrant to quantification⁴.

The approach has sparked interest within the scientific, societal, and political spheres^{6,7,8}. Defining the interconnectedness within bundles of boundaries makes it possible to articulate them so as to further clarify the issues at stake and give meaning in political and societal terms. **Here we show that the framework can be trimmed down to a two-component system: agriculture and physico-chemical deregulation (Figure 1.3).**

1. Agriculture is the main resource provider (labor) and consumer (land use change, biomass appropriation etc.) of all time. The corresponding land conversions constitute the “largest engineering project” in which mankind has ever engaged (Verburg et al, 2015). Agriculture and associated food systems⁹ concentrate the main stressors of planetary boundaries: changes in land and water systems, ecosystem alterations, fertilizer chemical flows, air and water pollutions, climate change. More specifically, agriculture entangles key biogeophysical interconnected building blocks of the biosphere - biomass, soil, and water. They are stranded assets in agriculture¹⁰ that need protection from environmental and management related risks through policy-decision making. This is important because the geopolitics of biomass, water, and land is driving the new resources scarcity agendas^{11,12} causing conflict between national security and international order instruments and mechanisms. In particular, land degradation has become a pervasive systemic process costing an estimated 10% of the annual global GDP and directly impacting the wellbeing of more than 3 billion people (IPBES report, 2018). The report indicates that actions to avoid, reduce, and reverse land degradation have costs three-times lower than the cost of inaction.

2. Global physico-chemical deregulation is a hidden global challenge. It is pervasive and systemic, being generated through ocean acidification, more general atmospheric, land, and water pollutions, waste accumulation, and climate change. Note that chemical intensification has been at work on a day-by-day basis over the last several decades: anthropogenic chemicals amount 144,000 distinct products, with a 500-fold

increase in volume and a 24-fold increase in assets (UNEP, 2012). This corresponds to a mega-pollution system consisting of complex cocktails that add to the natural chemical world¹ and change over space and time. That makes it hard, or even impossible, to measure, understand, and control a process in which limits - up to which organisms or ecosystems can safely cope with additive or multiplicative risks posed by the combination of multiple factors - have to be estimated with regard to exposure to any single factor. Literature survey of cocktail effects indicates that results are preliminary: additive to synergistic effects are observed in mixtures of 2-5 compounds in cultured cells or biomarker and genotoxicity response tests (Graillet et al, 2012).

A detailed knowledge of the response of the considered systems to all possible combinations of risk factors implies dividing the limit value for an individual threat by the square root of the number of threat dimensions (**Figure 1.4**). Even though this would be unrealistic in practice, it suggests that the adoption of limit values two or three order of magnitude smaller than those estimated by single factor impact studies¹ is relevant to legislation.

In the face of ongoing changing trajectories, it is unlikely for science to be able to deliver satisfactory answers in the near future because the physico-chemical deregulation is a wicked problem: lack of reference norms for cocktails operating since decades in evolving environments and bias in monitoring methods with improving sensitivity and accuracy¹. Furthermore, prevention protocols, standards, and legislation are hardly adapted to the scale of risks that remain largely unpredictable and even inexpressible.

Agriculture and physico-chemical deregulation act in synergy, illustrate the mis- and overuse of resources, and are first order systemic risks to planetary health² with long term effects on the society. For example, the annual economic costs of health impacts in food systems are estimated at \$13 trillion¹³; see also¹⁴ for estimates on pollution effects in terms of non-communicable diseases that kill 40 million people every year, with more than 80% in low income countries. The cost of inadequate action in preventing NCD is estimated at \$47 trillion in global GDP loss from 2011 to 2025. We argue that the ensemble constitutes the systemic great challenge with which all societal and political levels must engage over the next decade: *the health bubble*. However, while most problems and risks in food systems can be tackled, those embedded in the global physico-chemical deregulation appear more like *terra incognita* to science, politics, and society.

Thus, we call for coordinated measures targeting planetary health through:

- Coherent chemical simplification (effectively reduce, redesign, and recycle) in transitions to resource - sober and - efficient societies (see also¹⁵ on the Planetary Health Watch initiative). In particular, ban loading of novel chemicals in the biosphere. This also contributes to slowing down climate change;
- Shifting around “productivity first” by integrating sustainable agriculture practices into food system transitions⁹ with priority on land tenure, value, and restoration, land-use planning based on ecological landscape approaches (Rockström et al, 2017), and the food-health nexus (De Schutter, 2017);
- systematically and systemically addressing over-consumption, a stressor of both agriculture and physico-chemical deregulation, while reframing green insurance and investment policies in natural capital management.

These measures are at the apex of further changes in joint environmental and societal policies that articulate physical and social boundaries⁶ in a resource stewardship approach guided by simple, universal, and indivisible principles in support of a civilization contract^{2, 8}. The issue is to measure the total costs for society according to no action versus partial-to-complete phase-out of large sets of chemicals and unsustainable practices in order to align equity in health and wealth within the limits of the biosphere. The inclusive health concerns can dictate legal frames that enable fair access to and allocation of resources, a process for which the World Health Organization¹⁴ and the International Resource Panel² could ensure the initial coordination role.

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Chapter 2 General introduction of the Ecosystem Natural Capital Accounts (ENCA)

Outline

The chapter addresses four issues and aims at framing the ecosystem capital accounting according to its main characteristics:

- Introduction of the ENCA methodology in an accounting context by setting the scene of environmental accounting and defining the terminology requirements in the field;
- Makes a general presentation of the ENCA approach and its specific methodological requirements;
- Argues on the choice of the watershed scale for producing the first ENCA proof-of-concept;
- Reports on the changes and drivers of land cover stocks and flows, based on the CORINE Land Cover Geographic Database (CLC) for the years 2000, 2006, and 2012 and the corresponding land use changes for 2000-2006 and 2006-2012. The Rhône watershed surface data and its derivative Dominant Land Cover Types are stored in a geographic database in the *Pôle Scientifique de Modélisation Numérique* (PSMN) server to establish relationships with different environmental variables from the next chapters and the corresponding accounts.

2.1 Introduction

On terminology

Traditionally, accounting has the double purpose of assessing wealth and controlling the completeness and fairness of the books kept for that end. Indeed, correctly assessing wealth would it be in terms of disposable revenue or assets and liabilities implies recording all gains and losses.

As the term “accounts” refers to the tools used by policy makers and enterprises to evaluate their performances, it is often used out of context which creates damageable confusion. Accounts are particular sets of data and tables (accounting books) with precise characteristics. The first one is that they are established with the purpose of measuring the enrichment or impoverishment of the account holder. They are about values. This endogenous indicator is named “balancing item” to express that it is the result of fair and exhaustive recording of all receipts and costs, debits and credits, assets and liabilities. Due to their comprehensiveness, accounts need a common unit of measurement of values. Moreover, because they are tools not only for the holder but for a range of

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

stakeholders such as shareholders, clients and providers, trade unions, banks and other financiers, and the fiscal authorities, accounting statements need be standardised, as they are with the International Financial Reporting Standards for enterprises and the System of National Accounts (SNA) for the national accounts.

The System of Environmental-Economic Accounting – Central Framework (SEEA-CF) and the SEEA – Experimental Ecosystem Accounting (EEA) are, despite their name, not fully clear in this respect. They are based on a set of partial accounts which are not integrated as the SNA. For example, there is no definition of a common unit of measurement, except for a controversial possibility of a general accounting in money. Even in this case, the SEEA-CF limits valuation of assets depletion to subsoil assets and does not address ecosystem assets. The presentation in terms of assets itself limits the integrative character of the SEEA because assets are standing side by side and nothing being said on the way to aggregate them. For example, this is the case for forest and water, except considering their economic value (with the limitations previously mentioned).

The SEEA-EEA refers as well to “assets” which favours bits and pieces applications that do not match the systemic dimension of ecosystems. The SEEA-EEA is still experimental and solutions are still under evaluation in terms of ecosystem services assessment and valuation and finally natural assets valuation. A remarkable point is that both SEEA-CF and SEEA-EEA have the ambition of defining international statistical standards. There is however a risk of confusion as a statistical framework is not necessarily an accounting framework. For example, the DPSIR indicator to which refer partly statistical frameworks are not accounting frameworks. The former are collections of indicators or of statistical tables, while the latter are frameworks integrated according to very strict rules. Also, an accounting framework cannot be seen only at the statistical level: the modern national accounts have been developed on the basis of centuries of business and government accounting.

A particular mention should be made of a research carried out in the context of financial accounting standards. The CARE model developed by Rambaud and Richard (2015) aims at introducing natural capital depreciation in corporate accounting. “To this end, [they] analyse and (re-)define the concept of capital in an ecological accounting context.”. The CARE model states that “ontologies” (the measurement of physical degradation) are at the start of natural capital depreciation measurement for which restoration costs are considered as the main methodology. This approach is very close to that of ENCA, and does not correspond to that of the SEEA-EEA which is unclear on

ecosystem degradation and explicitly excludes restoration costs from the framework for the reason that it is not appropriate for measuring ecosystem services (UNSD, 2017).

Beyond these general approaches to environmental accounting, several methodologies should be considered as accounts on the grounds of the exhaustiveness of coverage of their realm, the existence of a common unit of measurement which allows aggregations, and suitability of defining a social or policy target.

The first one is Material Flow Accounting (MFA) (OECD, 2008). According to variants, all material flows can be reported in the framework. The common unit of measurement is the tonne and the policy target is to reduce the ratio Total Material Consumption/GDP.

The IPCC reporting framework to UNFCCC is a variant by some aspects of MFA. Better focused than MFA, it defines a common unit of measurement, the CO₂-equivalent which measures the contribution to the state of the atmosphere/climate system, defined in reference to the social target of mitigating warming increase to +1.5 degrees as compared to 1990.

Human Appropriation of Net Primary Production (HANPP; Haberl et al, 2007) is as well exhaustive and propose a common unit of measurement (the tonne of biomass) which allows aggregation. The reference to a climactic potential provides useful guidance regarding degradation and possibilities of restoration.

The Ecological Footprint (EF) Accounts, are formal exhaustive accounts with measurement of biocapacity converted to hectares. They highlight both national or local footprints and footprints embedded into international trade. However, beyond a general idea of reducing our consumption, the relation of the EF metrics to policy or social targets is not clearly defined (Van den Bergh and Grazi, 2014), which makes the EF a communication tool instead.

The approaches of ecosystem services valuation (see *Chapter 1* and *Suppl. Table 1*) use broadly the “accounting” term. It relies for part on the fact that they are all variants of Cost-Benefit Analysis (CBA) where the final decision requires valuing all the elements. This is called in CBA finding the “accounting price” for those elements which have no clear market price. However, many elements of accounting standards are missing. If a particular CBA is complete in its own case, it is difficult to aggregate the findings of several studies because of their contingency. The same issue exists for ecosystem services assessments which are often difficult to aggregate because of different measurement units. As mentioned previously, the monetary valuation solution is very controversial

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

as it implies a utilitarian approach of the natural capital and the acceptance of the low sustainability paradigm.

The use of the accounting terminology regarding the Planetary Boundaries set of indicators (Meyer and Newman, 2018) is equally misleading. These indicators are not formally connected and their transcription in DPSIR is certainly helpful but does not make the framework an account. It would be better to speak of “counts” rather than of accounts, which would help defining more clearly their status within the various approaches listed above.

Following the criteria outlined in the first paragraph, Ecosystem Natural Capital Accounts (ENCA QSP, Weber 2014) – an extension of SEEA-EEA (2012), is an integrated accounting framework which is relevant for any scale, from national accounts to local governments and enterprises accounting. It includes a common measurement unit (the Ecosystem Capability Unit, ECU) and most ecosystem services are recorded in ENCA QSP, in particular the provisioning ones, as flows of the ecosystem use table or as balancing items. For example, in the case of carbon sequestration what matters is the difference of positive and negative flows of the bio-carbon account. Uses of ecosystem services embedded into production and returns from the socio-economic system (e.g. CO₂ emissions, waste water, irrigation water) are recorded according to socio-economic statistics. The ECU allows integrating bio-carbon, water, and ecosystem integrity and biodiversity for the purpose of measuring ecosystem degradation, stability or enhancement and inform on the social target of “no net ecosystem degradation”. This corresponds to the “critical natural capital” variant of the strong sustainability paradigm.

The ENCA Landscape.

ENCA-QSP is an extension of the UN SEEA-EEA with the purpose of calculating ecosystem deterioration (or enhancement) and the liability of economic sectors to degradation. The ENCA-QSP accounting framework is based on a simplified model consisting of three interconnected and integrated components: bio-carbon, water and ecosystems infrastructure and biodiversity (**Figure 2.1**). The three components are interconnected by common levels of spatial representation used for defining the statistical units for which accounts are produced: the Socio-Ecological Landscape Units (SELU). Indeed SELU are defined as a combination of dominant land cover and physical properties such as belonging to a river sub-catchment and a class of altitude. Taking into accounts river basins boundaries allows integrating rivers and surface entities. In ENCA, SELU are the

geographical/statistical ecosystem units for which ecosystem capability, in terms of degradation, stability or enhancement can be meaningfully computed. Accounts interconnection encompass other dimensions such as the volume of soil for which organic carbon and humidity are estimated, the meteorological variables used for computing Net primary Production or the land cover map and account common to the bio-carbon and the ecosystem integrity and biodiversity accounts. **Figure 2.1** summarizes the spatial meshes used during this work to additionally illustrate the approach taken by the ENCA methodology.

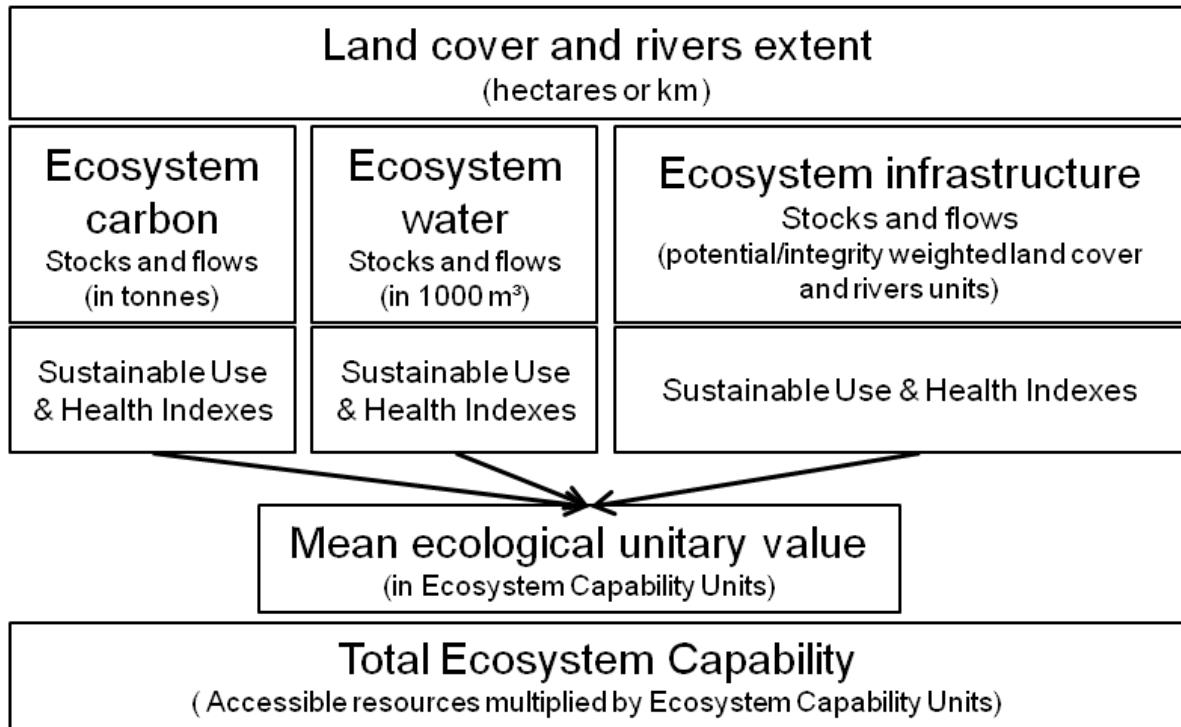
As for integration, ENCA-QSP relates formally to socio-economic and demographic statistics regarding the flows of resource use. The bio-carbon accounts details harvests of agriculture crops and timber by converting tonnes of products into tons of carbon. The water account records water uses as known through withdrawals statistics or estimated through population statistics multiplied by inhabitant-equivalents. One can note that bio-carbon accounts cover a large scope of the so-called provisioning ecosystem services. In addition, when compiling ecological debts, the exchanges between territories are reflected as impacts suffered or generated, including exchanges with the rest of the world. This question of exchanges is an essential aspect of water accounts.

Further on, internal integration of ENCA is provided by a structure common to the three components (**Figure 2.1**). Each account is made of tables of quantities which are the resource stocks and natural flows, the resource accessible without depletion and the use by economic sectors (including returns after withdrawals and use). Quantitative accounts deliver an index of intensity of use. For each account, an additional table records elements which are more difficult to quantify and called qualitative. These elements are used for making diagnosis of ecosystem health, summarized in an index. The indexes of intensity of use and of health are finally combined altogether to produce a new metrics of ecosystem ecological unit value, the Ecological Capability Unit. Each accessible resource measured in its own metrics is then multiplied by its ECU price to calculate its ecological value.

Being expressed now in the same unit, the total of the three resources and its change (degradation or enhancement) can ultimately be expressed in a single aggregate of ecological value. It requires however a last convention regarding the relative orders of magnitude of the measurement units. The choice should be made considering the importance of each component in the aggregate and the

consequences on its sensitivity to changes. For the three indexes of unit ecological value, the convention proposed in ENCA-QSP is to take them on par, with no weighting factor (Chapter 6).

Accounts for an ecosystem unit



ENCA-QSP: spatially defined statistical units for ecosystem accounting

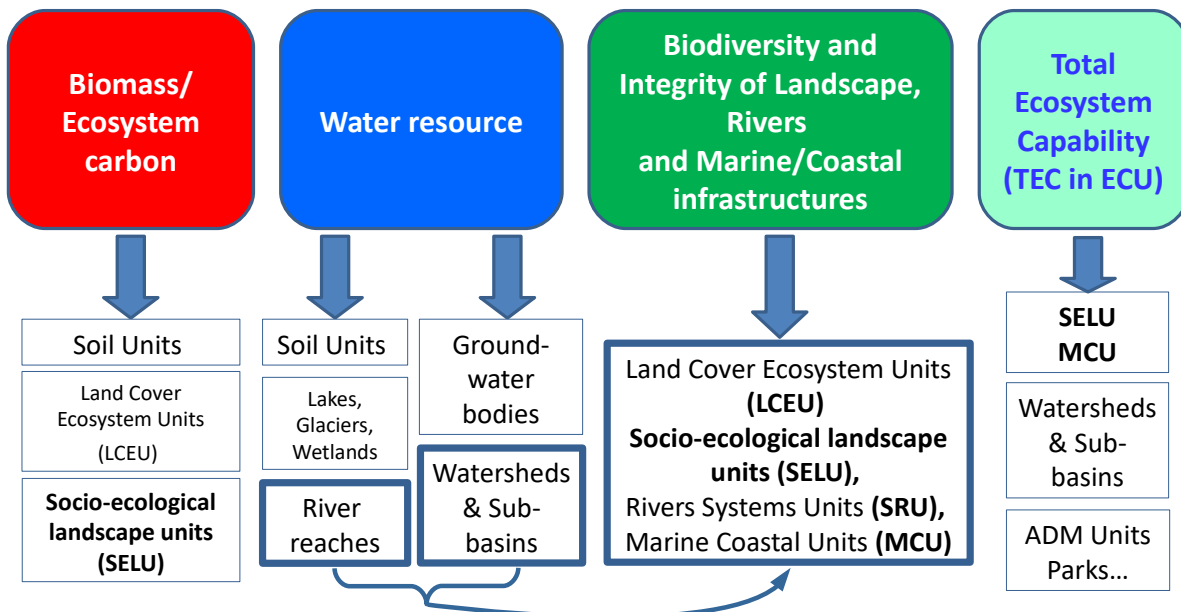


Figure 2.1. Top panel. Articulation and integration of ENCA components. Bottom panel. Diagram summarizing the different spatial meshes used to establish the accounts (ENCATS, UZHYD, SELU, administrative units) and illustrating the diversity of the

observation levels and the corresponding sources of data that have been mobilized during this work.

The purpose of ENCA is to quantify ecosystems with a common metrics in order to ultimately measure degradation (or improvement). Ecosystem degradation (or improvement) is considered as loss of ecosystem assets' ecological value. Ecological value differs from economic value as it does not consider the monetary benefits provided by the ecosystem services but the resilience of ecosystems functions. Economic benefits result inclusively from the combination of several types of capital: produced, human, social, cultural, and natural. In this perspective, aggregates of total or inclusive wealth cover possible substitutions between these types of capital, as acknowledged with the concept of "weak sustainability". Instead, considering the intrinsic ecological value of ecosystems restricts possibilities of substitution to the ecosystem capital itself. It is called "strong sustainability" as the total ecological value aims at being maintained. While economic value can be measured through observed market prices, there is no metrics for ecological value, no currency. ENCA proposes a simple model to compute a unit-equivalent for measuring ecological value, called "Ecosystem Capability Unit" (ECU). The model is based on three components: bio-carbon, water, and ecosystems infrastructure and combines quantitative and qualitative variables. The data infrastructure is based on the geographical characterization of land and rivers ecosystems. Quantitative tables record stocks and flows for measuring the resource accessible without depletion and compare it to the total use. Indexes of resource use sustainability are then combined with ecosystem health indexes to compute the average unit price in ECU for a given ecosystem unit. Each accessible resource can be multiplied by its ECU coefficient. Being expressed now in the same unit, the three resources can be added up to calculate the headline indicator of ENCA: Total Ecosystem Capability, and its change, degradation or enhancement (Weber, 2014).

The corresponding calculations require significant data collection, assimilation and integration, including data derived from Earth observation by satellite and other maps, meteorological and hydrological data, soil maps, biodiversity monitoring data, agriculture and forestry statistics, population censuses, or administrative registries.

Each chapter deals with one account of ENCA. I start here with the land account as it is the key infrastructure for all accounts.

The ENCA approach is articulated through the land cover and rivers frame.

The ENCA accounts are introduced by and are framed in the land accounts. They describe the dynamics of land-cover and use, how resource stocks change over time to identify patterns and trends, and to understand better the implication of those changes for land management (Haines-Young et al., 2006). This account answers questions about the maintenance of the quality and integrity of the land stock, and the optimal compensation for the land-cover changes.

The land-cover accounts are also the foundations for the carbon, water, and infrastructure components. They serve to structure the whole accounts, and to define and implement the statistical units for accounting (Weber, 2014).

On these lines and more generally, land use patterns constitute an extremely important issue *per se*. The process of land use change – the largest geoengineering human activity of all times (Verburg et

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

al, 2015), is the main driver of water, biomass, carbon, and ecosystem fragmentation (with habitat loss and degradation, and biodiversity erosion) (Barnosky *et al*, 2012; Urban, 2015; Steffen *et al*, 2015; Verburg *et al*, 2016). Among the consequences, one should mention the reduction of ecosystem productivity and climate change.

Beyond the high levels of human appropriation of biomass, what matters is the rate and extent of alterations produced through land use changes and accelerating fragmentation of landscapes and habitats. Deforestation (mainly in the tropics during the last century), grassland burning, and permanent cropping displace natural ecosystems and reduce primary productivity of the new land cover. For example, converting forests to farmland resulted in replacing ecosystems whose aboveground phytomass was 50-100 tC per ha with cropping typically yielding below 10tC per ha (Smil, 2013). In addition to effects on bioproductivity, there are second-order effects on ecosystem productivity, such as change in micro- and meso-climates (CO₂ emissions and other air and water-born pollutions), evapotranspiration and soil moisture, soil erosion rates and fertility reduction / loss, pathogen and invasive species impacts etc. (Smil, 2013).

Concerning soil degradation, the political and socio-economic drivers of erosion continue to be neglected. For example, land tenure is part of the equation. There is a large disconnect between the financial value and the value of land according to multifunctional capabilities of land (Thomas and Schauer, 2016). This is because land provides food, income, and many other amenities that are critical for both vulnerable individuals and communities and countries in terms of food, health, and energy security.

The ENCA approach is performed at the scale of the watershed. The experimental work reported in this PhD research is dedicated to the Rhône watershed. Watersheds, or water catchments or river basins, are considered important conceptual frameworks and natural systems in which to investigate complex socio-ecological processes (Parkes *et al*, 2010; Jenkins *et al*, 2018). They are functionally coherent hierarchical networks that can mobilize social and territorial actors and institutions located within their boundaries through shared common history of social and environmental issues, such as land use policies, water governance, and ecosystem management. That context of interdisciplinary and local knowledge and know-how is the foundation of this project, aiming at, as stated by Tulmin (2017), “building better and more resilient connections between institutions and ecological resources. All too often, vital ecological resources are split by administrative boundaries which make nonsense of the natural landscape. This is especially the case with rivers and wider

watersheds, where the geographic integrity of the river basin is rarely matched by an administrative system with the powers required to manage upstream-downstream interactions”.

2.2 Definition of statistic accounting units

The ENCA approach makes use of statistical or analytical units. While national accounts are essentially legal in nature, the accounting units in ENCA are essentially geographical and are defined from biophysical characteristics for each of the accounts. The statistical units are spatial areas in which information is collected and statistics are compiled (SEEA-EEA, 2012). From land cover data the Dominant Land Cover Types and Socio-Ecological Landscape Units (SELUs) were defined (see Results below, the Carbon account **Chapter 3** section 3.2.4 and **Figure 3.13**, and the Water account **Chapter 4** section 4.2.3).

2.3 Land cover stocks and flows

The stocks of land cover correspond to the physical areas of different land cover types. The flow accounts include the information about how the land cover changes were produced, i.e. the transference or exchange between land cover categories in a certain period. The land-cover flows are organized in a transition matrix according to the initial and final land-cover type, allowing to generate all possible combinations. The classification of the flows explains the type of change between land cover types (i.e. artificial development over agriculture).

The flow accounts include the losses of initial land-cover for each land cover type, called consumption, the creation of new areas, called formation, and the balance between consumption and formation.

2.3.1 Methods

The Rhône river watershed is a transboundary basin shared by France and Switzerland. This area encompasses several valleys and rivers in three major regions in Europe, alpine, continental, and Mediterranean (Olivier et al., 2009); five administrative regions and 26 departments in the French part, and three cantons Vaud, Valais, and Geneva in the Swiss part.

Land Cover Map datasets (CLC) from the Coordination of Information on the Environment (CORINE, 2017) were used for the analysis on land cover and change. The CLC consists of an inventory of land cover in 44 classes complemented by change layers, which highlight changes in land cover. This is produced by visual interpretation of high resolution satellite imagery providing consistent information on land cover and land cover changes across Europe. The CLC (1-ha pixel-size) on land cover for the years 2000, 2006 and 2012, and change layers were clipped using the Rhône basin surface (see Water chapter) as a mask (Appendix Aⁱ) in the *Pôle Scientifique de Modélisation Numérique* (PSMN) server. They were sent to the SQL spatial database, in the same server, called “eccounts_rho” with a bash script (Appendix Bⁱ). The accounts were done using the 44 land cover classes at the third level of the CLC and 28 classes of flow (Weber, 2014).

The area of the stocks was calculated using query 1 (Appendix Cⁱ) by counting the number of pixels of each land cover class. The area of the flows was calculated with query 2 (Appendix Dⁱ) by counting the number of pixels by flow class from 2000 to 2006 and from 2006 to 2012.

ⁱ Appendixes: List of computer scripts. Available from : <https://drive.google.com/open?id=1obTHtMKIS0XVt0qoA7WOx7XjDmmxvK63>

Box 2.1. Software Packages

The ENCA-QSP tests have been carried out using the following software packages according to the best convenience of use for specific tasks:

QGIS and SAGA-GIS for geographical information processing. These two packages are partly integrated and their main data formats are fully compatible. They are .shp (the ESRI shapefile format), .tiff and .sdat (the SAGA grid format, read by QGIS). These are two powerful freeware packages fit for the purpose of accounting. QGIS is particularly fit for cartographic work with a powerful shape files editor. SAGA targets scientific calculation needs, with a library of circa 500 modules. De facto, ENCA can be implemented with other GIS packages, e.g. with ArcGis

SAGA-GIS: System for Automated Geoscientific Analyses

<http://www.saga-gis.org/en/index.html>

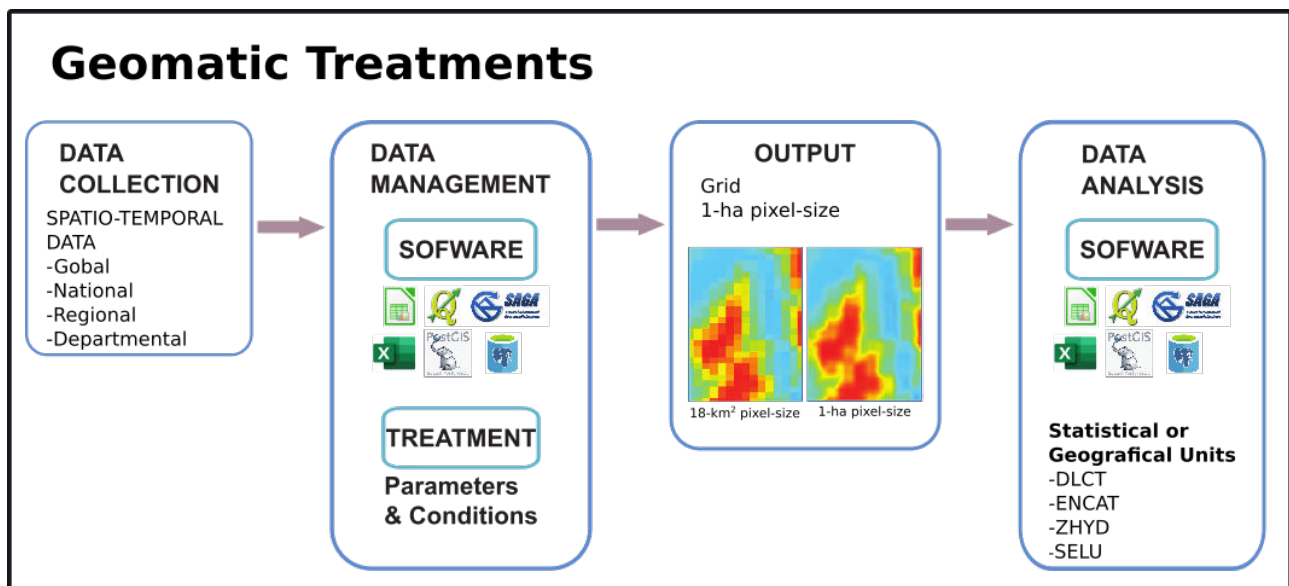
Ref. : Conrad, O., Bechtel, B., Bock, M., Dietrich, H., Fischer, E., Gerlitz, L., Wehberg, J., Wichmann, V., and Böhner, J. (2015): System for Automated Geoscientific Analyses (SAGA) v. 2.1.4, Geosci. Model Dev., 8, 1991-2007, doi:10.5194/gmd-8-1991-2015.

QGIS : <https://qgis.org/en/site/>

MS Excel and LibreOffice Calc have been used for integrating accounts. Their formats are fully compatible. Final integration and data management has been carried out with PostgreSQL and PostGIS.

PostGIS for geographic objects to the PostgreSQL object-relational database : <https://postgis.net/>

Box 2.2. Schematics of the production of grids with same 1-ha pixel-size for data analysis



2.3.2 Results

The study area covered 97,456.80 km². The analysis of main categories of CORINE (2017) datasets showed that forest, and pastures and mixed agricultural areas are the dominant land covers in the watershed, 36% and 23%, respectively. They are followed by arable land, natural grassland, open spaces, artificial land, transitional forest-shrub, wetlands and water bodies, and glaciers (Figure 2.2).

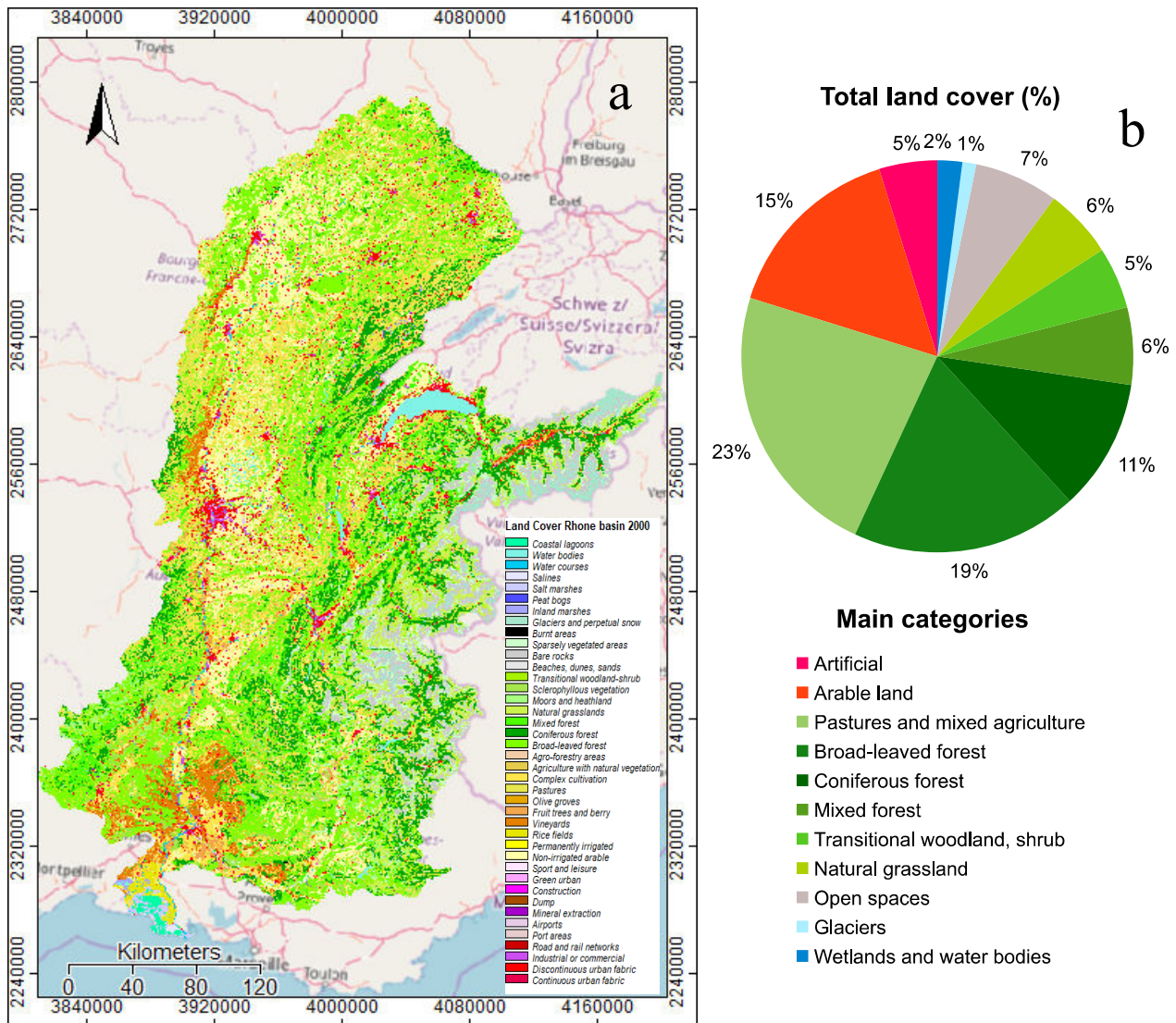


Figure 2.2. Land-cover from CORINE Land Cover in the Rhône watershed representing 40 land cover types (a) and percentage for grouping them in main categories (b) using the same logic for coloring as in (a). The observed changes for the period 2000-2006-2012 were relatively small and did not change the percentage of the main categories.

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

For the analysed period, from 2000 to 2012, glaciers decreased by 13.65% (149 km²), natural grassland by 2.04 % (114.64 km²), and pastures and mixed agricultural areas by 1.36% (304.6 km²). Artificial areas increased by 10.64% from, 4,666.43 km² to 5,163.13 km² (**Figure 2.3**). To put the latter in perspective, artificial areas grew by an area roughly the size of Grand Lyon.

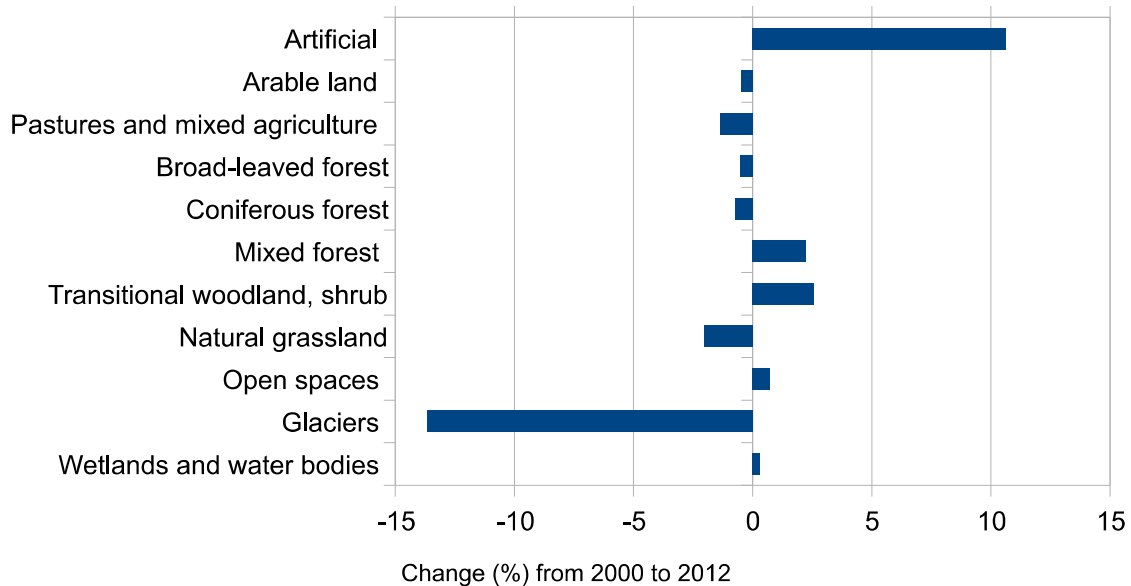


Figure 2.3. Land cover change percentage from 2000 to 2012. The changes are given relative to the individual area of each class in 2000. The analysis was performed in CLC as a postGIS object using PostgreSQL queries.

The total changes for the first period, from 2000 to 2006, were 391.55 km². The net changes (formation minus consumption) showed increase in artificial land, transitional woodland, open spaces, and wetland and water bodies; and decrease in agricultural and most of natural surfaces (**Figure 2.4**). Overall, the trend is some degradation of ecosystem quality.

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

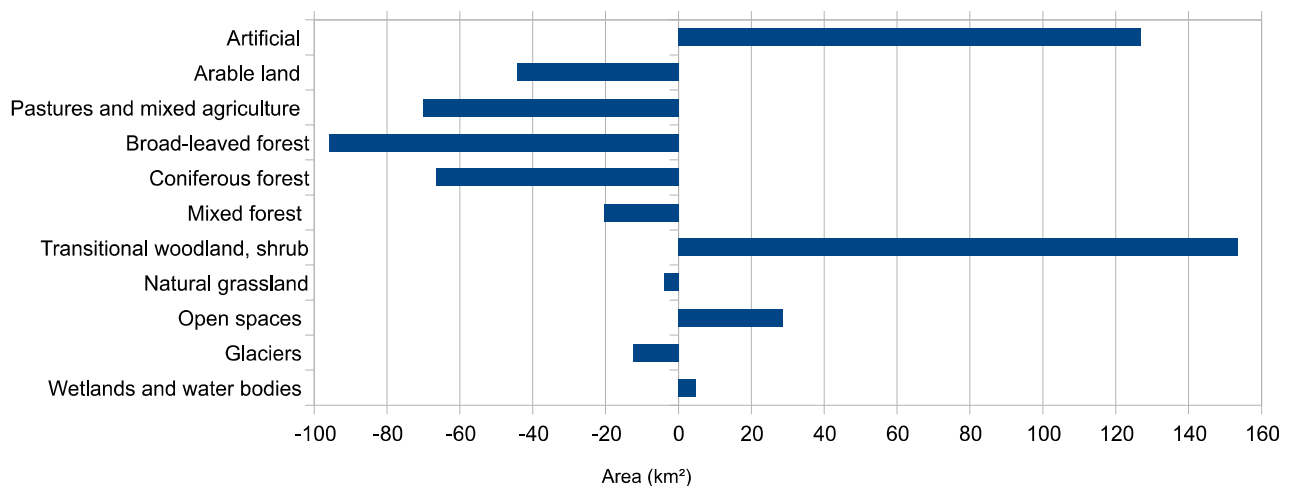


Figure 2.4. Absolute net changes from 2000 to 2006. Resulting from consumption and formation of the individual classes. The analysis was performed in CLC as a postGIS object using PostgreSQL queries.

The drivers of these changes are mainly artificial development and forest degradation. The increase of transitional woodland shrub is explained by the degradation of forested areas. The increase of artificial land was principally due to the transformation of cropland (**Figure 2.5**). The latter is likely to represent lost of fertile agricultural land mainly due to land artificialization.

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

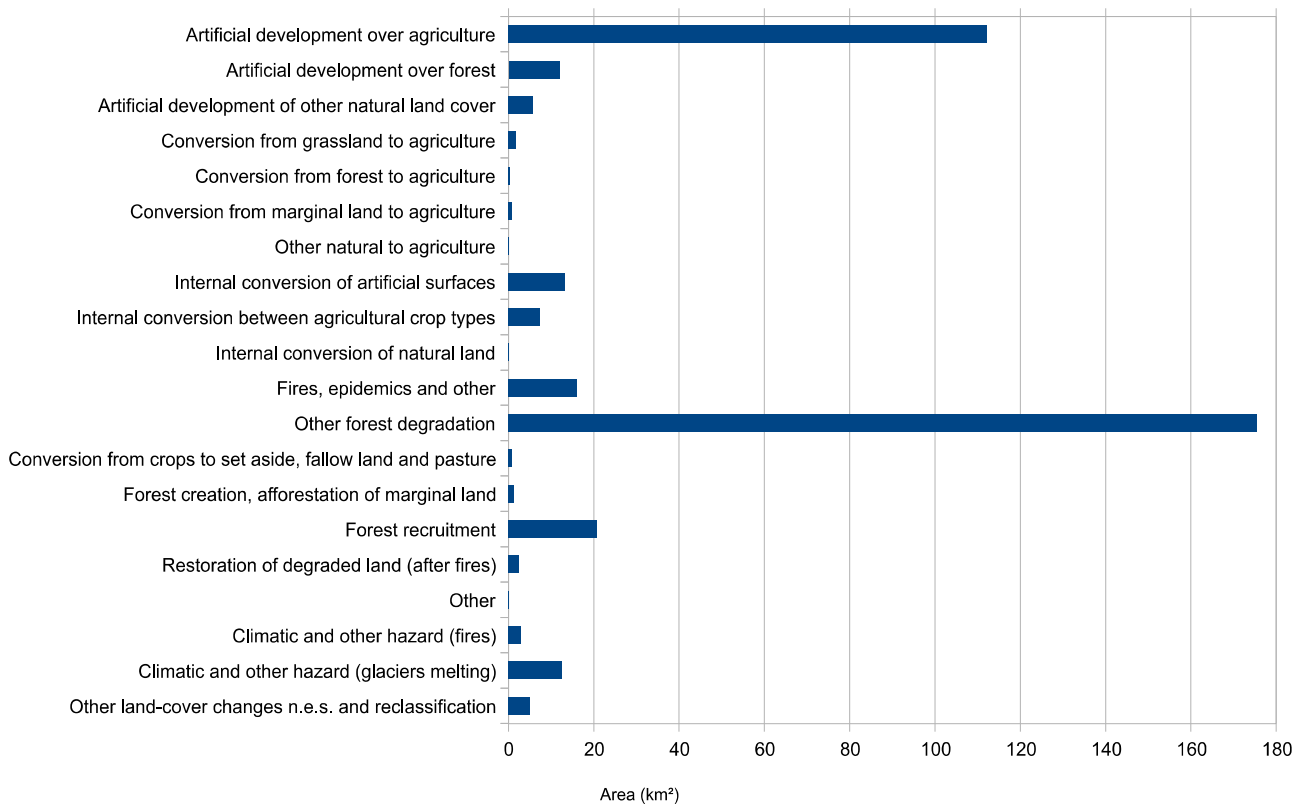


Figure 2.5. Drivers of land-cover changes from 2000 to 2006. The n.e.s. initials stand for not elsewhere specified. The analysis was performed in CLC as a postGIS object using PostgreSQL queries. The drivers of change were classified according to the type of change between the two years (Weber, 2014).

For the second period, from 2006 to 2012, a total of 454.00 km² underwent change. The net changes correspond to increase in artificial, transitional woodland shrub, and open spaces; and decrease in agricultural and most of natural surfaces (**Figure 2.6**).

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

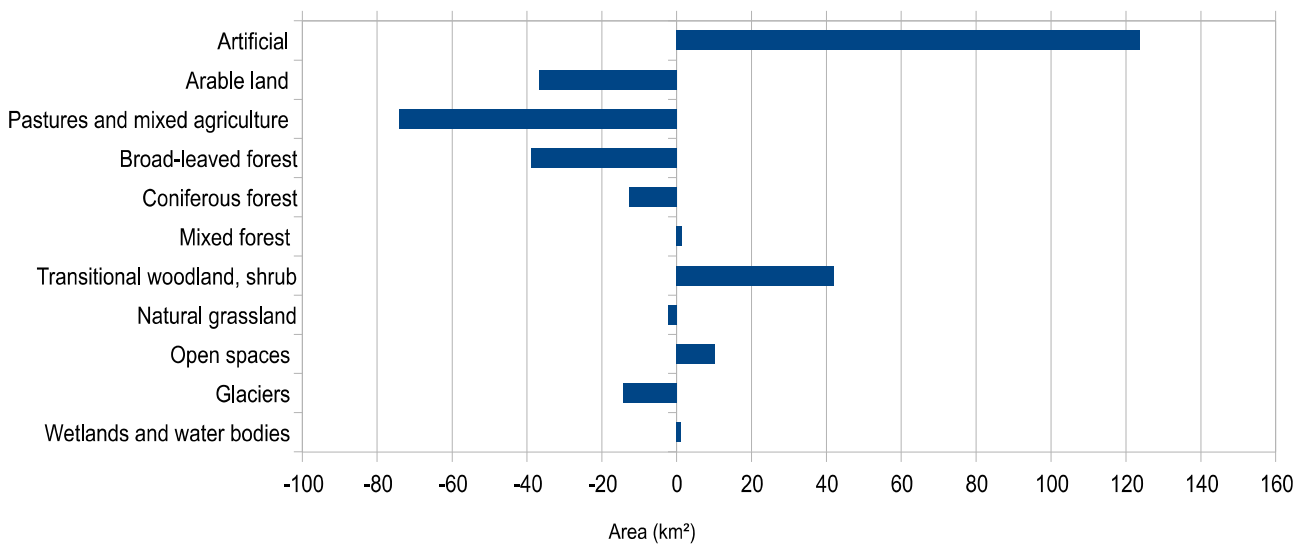


Figure 2.6. Absolute net changes from 2006 to 2012. Resulting from consumption and formation of the individual classes. The analysis was performed in CLC as a postGIS object using PostgreSQL queries.

The main changes resulted from the transformation of agricultural to artificial land, the degradation of forest to transitional woodland shrub, and the recruitment from transitional woodland shrub to forest (**Figure 2.7**).

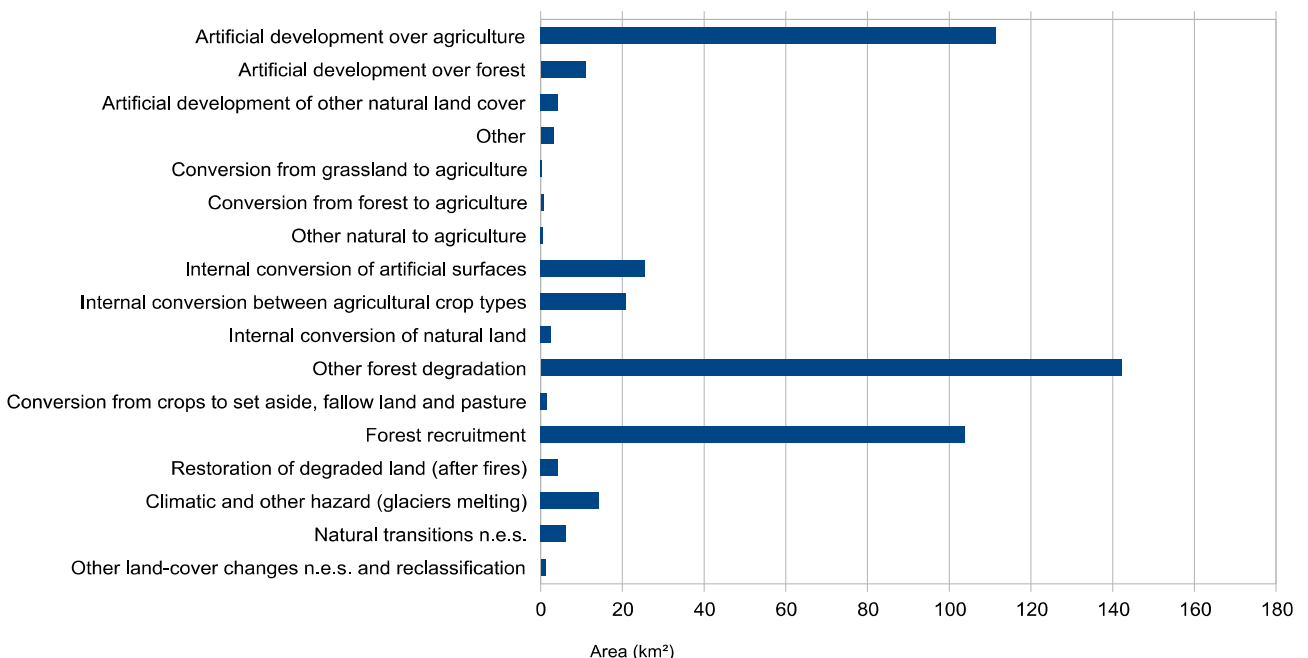


Figure 2.7. Drivers of changes from 2006 to 2012. The n.e.s. initials stand for not elsewhere specified. The analysis was performed in CLC as a postGIS object using PostgreSQL queries. The drivers of change were classified according to the type of change between the two years (Weber, 2014).

2.3.3 Main issues

The grouping of stocks and flows categories should be made carefully to avoid misclassification and misinterpretation. For example: considering only one category for forest creation and management, as in Haines-Young et al., (2006), gives confusing results because transitional forest could result either by consuming forest or by forest succession or regeneration. In this example, 97.83% of the 179.37 km² in the period 2000 to 2006 and 59.31% of the 250.56 km² in 2006 to 2012 of the transitional woodland shrub created are due to forest degradation which is not evident in the graph (Figure 2.8).

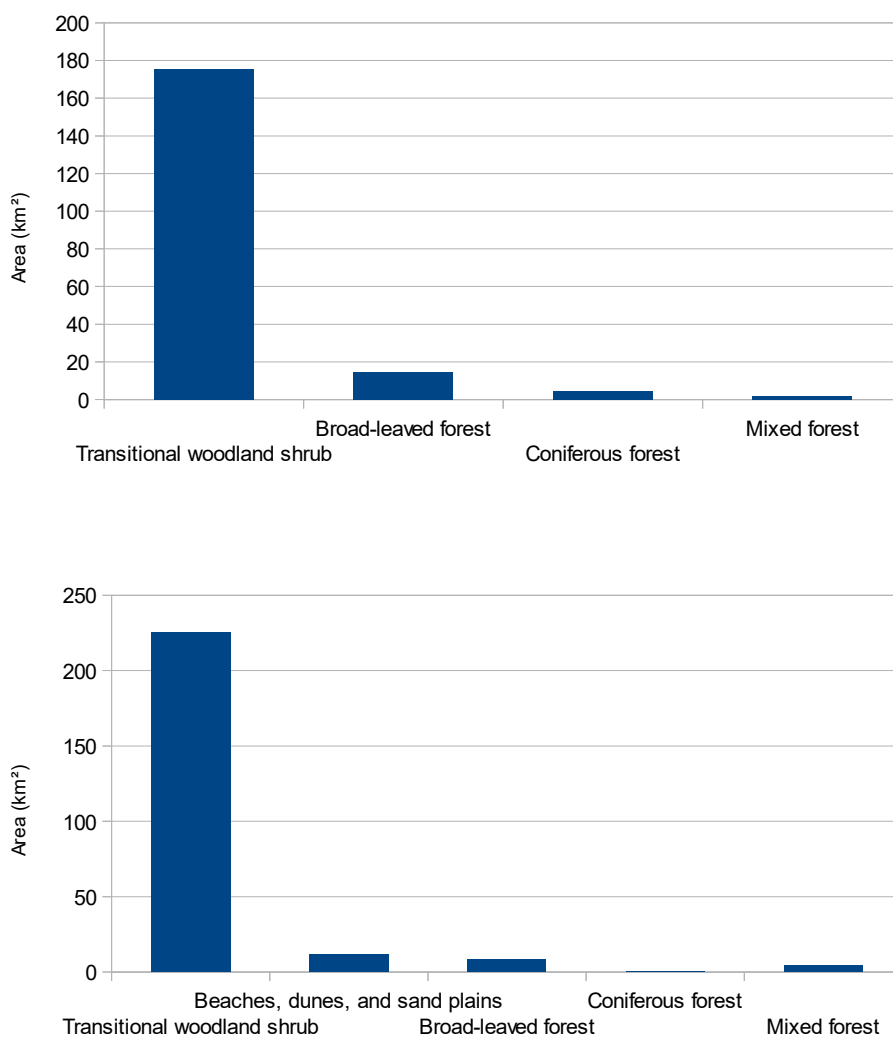


Figure 2.8. Forest creation and management category 2000 to 2006 (top) and 2006 to 2012 (bottom). The transitional woodland and shrub could result either by consuming forest or by forest succession or regeneration.

Chapter 2 Introduction of the Ecosystem Natural Capital Accounts (ENCA)

Another issue is properly designing flow classes according to dominant vegetation types of the area of interest. In this classification, fires are considered to impact only forest, while in the Rhône watershed from 2000 to 2006 natural grassland and sclerophyllus vegetation also experienced fire. Furthermore, keeping consistency with other classifications, such as RAMSAR, requires grouping classes in ways that can produce loss of information. For example, wetlands alone decreased by 1.30% in 2000 to 2006, but the information is lost by integrating wetlands together with the water bodies class.

Concerning the CLC, the resolution of the maps is different between years and between the maps of land cover change resulting in differences in surfaces in stocks and flows when balance sheets are used.

Land cover and land-use change analysis allows studying important aspects of the state of terrestrial ecosystems. This is a necessary but not sufficient element in understanding the capability of land to maintain the full range of ecological processes and functions (MEGS, 2013), as I will show in the next chapters.

Conclusion

In the first part of this Chapter, ENCA was defined as one of the accounting frameworks that is exhaustive and integrates the value of the ecosystem in a common aggregated measurement unit called “Ecosystem Capability Unit, or ECU”. This unit allows the integration of bio-carbon, water, and infrastructure to measure ecosystem degradation, stability or enhancement, which has not been achieved so far with other environmental approaches analysed in Chapter 1.

The results on land cover change generated in the second part of the Chapter indicate that Forest, and Pastures and Mixed Agricultural areas are dominant land cover types in the Rhône river watershed. The main changes over a 12 year analysis show that Glaciers decreased by 13% and Artificial areas increased by 10%. The main drivers of change in land cover were the Artificial land expansion over Agriculture and Forest degradation. The results will serve to calculate the loss of carbon due to land change in the next chapter, Chapter 3. Also land cover datasets serve to describe and quantify the interactions between economy and nature by providing: statistical units for data integration of the bio-carbon, water, and ecosystem infrastructure accounts, localization, and change monitoring.

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Chapter 3 Ecosystem Carbon Account

Outline

The chapter assesses the state of the bio-carbon resource (tC ha⁻¹), one of the three core accounts in the ENCA methodology. Of note, bio-carbon data sources represent a combination of satellite, *in situ*, and census-based production of information. Ten variables on stock, flow, and use are analysed as well as the loss of bio-carbon due to land cover change. The index of intensity of use is calculated as the ratio between the Accessible resource in bio-carbon and its use.

3.1 Introduction

Changes to ecosystem functions and derived services are faster than ever in human history (Millennium Ecosystem Assessment, 2005) essentially due to human population increase, over-consumption of resources, and technology-driven ecosystem use intensification (Carpenter et al., 2009).

Governments or national, regional, and local companies do not keep systematic natural capital consumption records. This consumption corresponds to the loss of ecosystems' ability to provide goods and services and is equivalent to creating ecological debts, a burden to future generations or to countries whose products are manufactured under unsustainable conditions (Weber, 2012). The accumulation of ecological debts represent economic and political risks and are a matter of national security and sovereignty.

Accounting for natural capital degradation allows adapting available resources to the basic needs of populations and maintaining of life-supporting natural systems on which societies depend on. A range of environmental evaluation tools have been developed with the purpose of integrating the natural capital into economical national accounting frameworks and inform policy-makers on their availability, use, and depletion.

Three methodological categories are presently in development: reference value or boundary indicators; ecosystem services valuations; system approach tools.

My work concentrated on the system approach methodology, the Ecosystem Natural Capital Accounting (ENCA). ENCA is an extension of the conceptual UN Economic and Environmental Accounting System (SEEA-EEA, 2012), with the purpose of calculating ecosystem degradation (or

enhancement) at different spatial scales. Importantly ENCA targets “no net ecosystem degradation” and ultimately the amortization of the ecosystem capital.

The ENCA protocol is based on three distinct accounts: bio-carbon, water, and ecosystems infrastructure and biodiversity which require significant data collection and integration, including available satellite imagery, standard coefficients, official statistics, and surveys. As far as possible data are converted to grids (rasters) of the same pixel size to facilitate the calculations needed for the accounting. Each account is made of tables of quantities: resource stocks and natural flows, the resource accessible without depletion, and the use by economic sectors (including returns after use). Quantitative accounts deliver an index of intensity of use. For each account, an additional table records qualitative elements. These elements are used for making diagnosis of ecosystem health, summarized in an index. The indexes of intensity of use and of health are combined altogether to produce the Ecological Capability Unit. The three entities are added up to calculate the asset value and its change, degradation, *status quo* or enhancement (Weber, 2014; **Figure 3.1**).

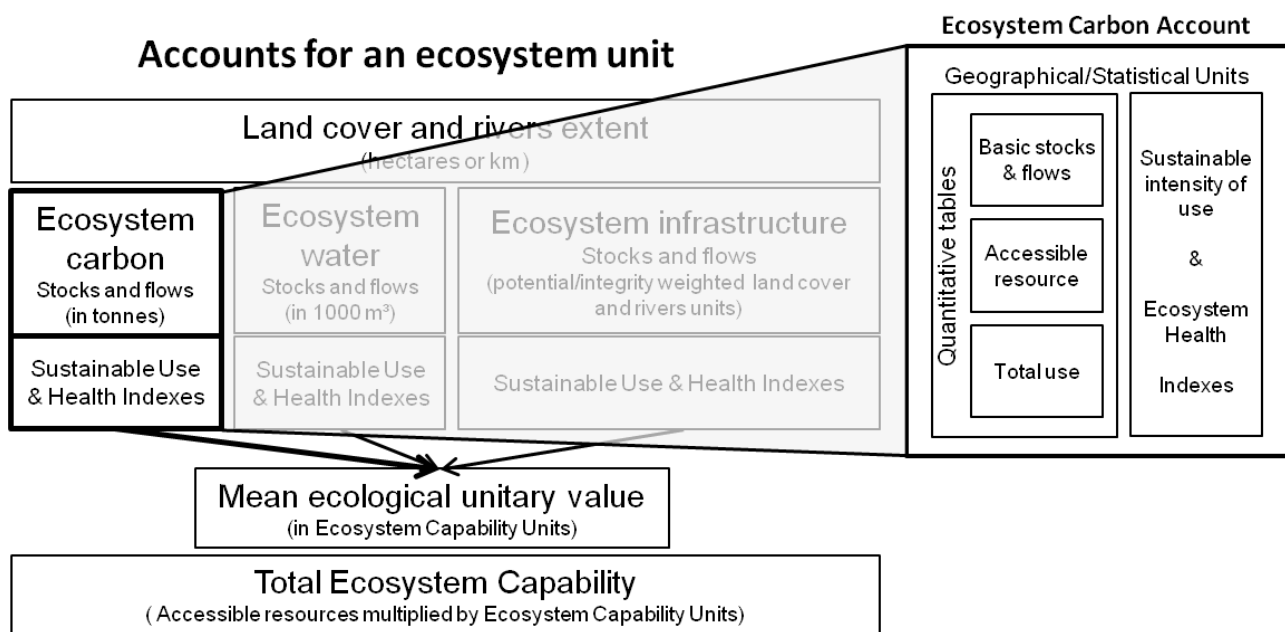


Figure 3.1. Articulation and integration of ENCA components. The model is based on three components: bio-carbon, water, and ecosystems infrastructure and combines quantitative and qualitative variables. In this chapter, I report the Ecosystem carbon account (rectangles in bold). Quantitative tables record stocks and flows for measuring the resource accessible without depletion and compare it to the total use. The corresponding calculations require significant data collection, assimilation and integration, including data derived from Earth observation by satellite and other maps, meteorological and hydrological data, soil maps, agriculture and forestry statistics, population censuses, or administrative registries.

I will present data on carbon account in view of their subsequent aggregation in ECU. I will discuss the strengths and limitations of the methodology in view of its optimization and user-friendly exploitation.

In the ENCA methodology bio-carbon is measured through an ecosystem approach. The carbon account records the ecosystem's sustainable capacity to produce biomass, expressed as bio-carbon based on conversion formulas. The biomass is a measure of the quantity of living matter and the resource of food, materials, and energy, as well as climate-biosphere interactions (Weber 2014). No aggregation of carbon accounts on stock, flows, and use in bio-physical values have been reported so far. The multi-disciplinary research on carbon accounting explores the subject in discrete inputs and outputs (see Nature, 2018). For example, Canada ecosystem accounting (MEGS, 2013) based on the SEEA Experimental Ecosystem Accounting (United Nations, 2013) reported the biomass extraction for human use from Canada's terrestrial and aquatic ecosystems for 2010. Other carbon accounting systems have been conceived to evaluate emissions (IPCC, 2006) and research on global carbon budget focuses on the perturbation of the global carbon cycle caused by anthropogenic activities and the increase of CO₂ in the atmosphere (Le Quéré et al., 2018).

3.2 Methods

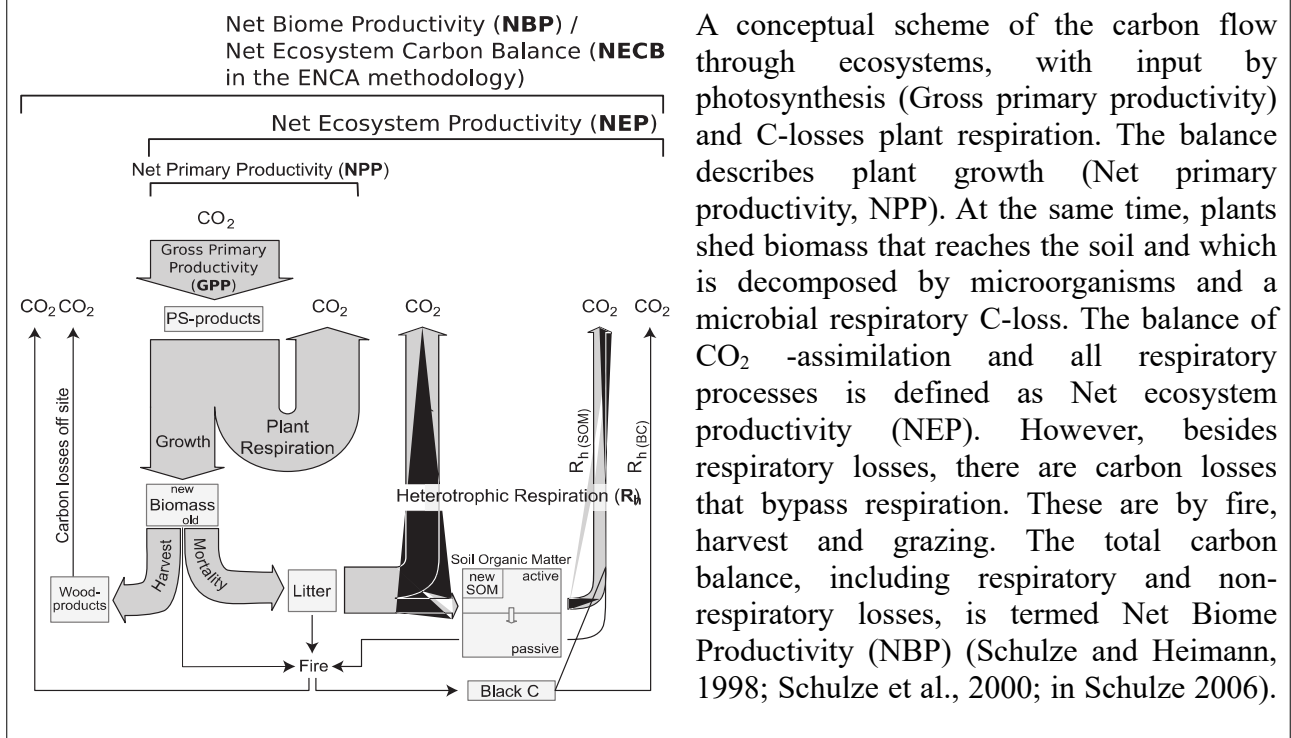
Box 3.1. Notions and definitions used to produce the bio-carbon accounts.

Biomass is the total mass of living matter within a given unit of environmental area (de Bosseoreille de Ribou et al, 2013). In agriculture, biomass production is called yield (biomass x land area) expressed in amount per hectare.

([http://www.footprintnetwork.org/en/index.php/GFN/page/glossary/#biocapacity];
[http://wwf.panda.org/about_our_earth/all_publications/living_planet_report/2010_lpr/])

Primary production is the biomass produced by plants per unit area, expressed in either units of energy [joules/(m²)] or dry organic matter [kg/(m²)] and its carbon content (gC m⁻² or tonnes C ha⁻¹). The transformation coefficients are: 1g dry biomass = 5 kcal = 0.5 g equivalent C (Barbault and Weber, 2010).

The respiration load reflects the autotrophic growth and maintenance costs in photosynthetic primary producers. Such costs average half of the gross primary production (GPP). The rates vary from 0.3 to 0.85, being highest in tropical rain forests (Smil, 2013). The remaining biomass is feeding the food networks in the biosphere.



The stocks of biomass produced by nature are partly extracted for human use and enters the economic system. It subsequently returns to nature as greenhouse gas, sludge or solid waste. In addition, humans may disrupt the carbon cycle by fire, erosion, or changes in land use, to name a few disturbances. The addition or subtraction of these flows from the net ecosystem production gives the net ecosystem carbon balance. The net ecosystem carbon balance can be calculated either

as the difference between inflows and outflows, or between opening and closing stocks. The opening stock of bio-carbon corresponds to the standing biomass or the quantity of bio-carbon in each reservoir (i.e., soil, vegetation, and water), the inflow of bio-carbon is the production of biomass, and the outflow of bio-carbon is its use (Weber, 2014).

The ecosystem carbon account structure consists in a set of four tables, the basic balance of stocks and flows, the use of the carbon, the accessible resource surplus, and indexes of intensity of use and of ecosystem health/distress. The last two tables are calculated with the data of the first two.

The main accounting items of the stocks, flows, and use for ENCA methodology are:

- Bio-carbon in aboveground living biomass
- Bio-carbon in litter and deadwood
- Bio-carbon in soil
- Other bio-carbon pools
- Gross primary production
- Net primary production
- Heterotrophic respiration
- Net increase of secondary bio-carbon stocks
- Agriculture harvested crops
- Wood removals
- Consumption and production returns
- Withdrawal of secondary bio-carbon
- Net indirect anthropogenic losses of bio-carbon and biofuel combustion
- Outflows

The data sets for 10 of the 14 main accounting items were converted to grids (raster) of the same size and same 1-ha pixel-size. They defined the Rhône grid system. Most of the accessible data sources run up to various periods of time and eventually correspond to one sampling time.

3.2.1 Stocks

Bio-carbon in aboveground living biomass: Living trees tC ha⁻¹

This carbon stock corresponds to plant biomass at the surface, particularly to the volume of timber (Weber, 2014). The carbon tonnes of trees by hectare were calculated using the volume of trees and area (m³ ha⁻¹) from the forest inventory at department level (IGN, 1992-2004), the vector database of forested polygons from the IGN (2006-2017), the tree relative cover in the watershed from Hansen et al., (2013), and conversion coefficients to biomass (dry weight) and to carbon in biomass from the IPCC (2003; 2006).

Data analysis

The forest inventory data was downloaded selecting each of the 26 departments that make up the Rhône basin, the last inventory campaign, the production forest, the area, and the volume. The series of data for the period 2000 to 2012 (**Figure 3.2**) were estimated from the inventory and using an average annual growth of 1% (Weber, 2016). The volume of the 23 tree species was divided by four to obtain the tonnes of carbon based on the assumption that the density of timber is in average 0.5 tonnes m⁻³ (IPCC, 2003) and the proportion of carbon in the dry biomass can be calculated with a factor of 0.5 (IPCC, 2006). Thus:

$$tC = tree\ volume \times 0.5 \times 0.5 = tree\ volume \div 4$$

The result was divided by the area to obtain the tonnes of carbon per hectare (tC ha⁻¹).

A correspondence table was created to match the records from the two sources. To join the forest inventory data (23 species) with the polygon data (19 species), certain species were grouped according to the IGN classes, and the geometric mean of tonnes of carbon per hectare was calculated. When there was no such information for a given department, tree species were assigned the geometric mean of the same species from other departments or of all the species.

Statistics spatialization

The files of the polygons for 26 departments were reprojected to the ETRS89 Lambert Azimuthal Equal-Area projection in QGIS. The new field “code_dep” in the attribute table was created and the department code was informed using the QGIS *field calculator*. The two files lacking the attribute *essence* (tree species) were reclassified with the attribute *LIBELLE* according to its correspondence to the *essence* (**Appendix Iaⁱ and Ibⁱ**). The files were merged using SAGA-GIS *Merge vector layers*, the surfaces other than forest were selected and eliminated, and a code was given to the *essence* category (**Appendix Ic**).

The $tC \text{ ha}^{-1}$ by species and department were joined in QGIS to their corresponding spatial unit using an identifier code consisting of the species and departments codes.

Estimation of carbon in trees outside the forest

The surfaces covered by trees outside forested areas were obtained using *Symmetrical Difference* from QGIS geoprocessing tools in the merged forested areas. The new polygons were transformed to raster (SAGA-GIS *Shapes to Grid, data / no data*) and multiplied using SAGA-GIS *Grid Calculator* by the tree cover from Hansen et al. (2013) interpolated to 1 ha in SAGA-GIS *Resampling Bilinear interpolation*. The result was multiplied in SAGA-GIS *Grid Calculator* by the carbon outside forested areas. It was calculated as the mean timber volume produced outside the forest ($28\text{m}^3 \text{ ha}^{-1}$; FAO FR from Weber, 2016). Thus:

$$tC = \text{tree volume} \times 0.5 \times 0.5 = \text{tree volume} \div 4$$

The resulting carbon in trees of the forest and outside the forest was merged in SAGA-GIS

Mosaicking (Figure 3.2).

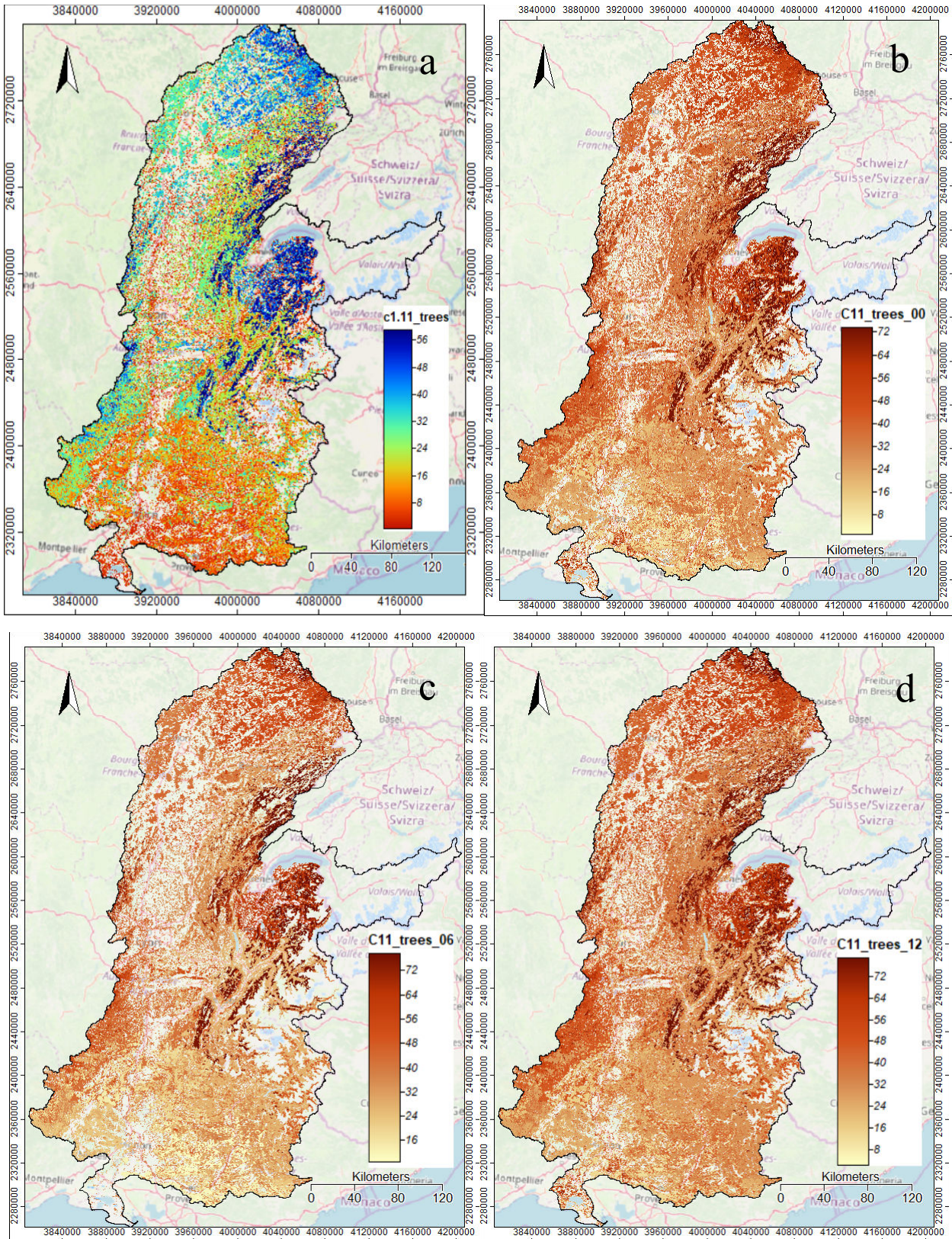


Figure 3.2. Bio-carbon ($tC\ ha^{-1}$) in aboveground living biomass. Spatialization of data on living trees from the *Inventaire Forestière Nationale* at French department level, using QGIS and SAGA-GIS geo-processing. Cumulative and integrated information by department reported in different

years between 1992 and 2004 (a). Estimations for 2000 (b), 2006 (c), 2012 (d) assuming an annual biomass growth rate of 1%.

Bio-carbon in litter and deadwood

The estimation of this carbon stock corresponds only to dead-standing trees and wind-throw in forested areas. The carbon tonnes by hectare were calculated from the volume and area ($\text{m}^3 \text{ha}^{-1}$) of the forest inventory at regional level (IGN, 2011-2015), the spatial data of forested area from the IGN (2006-2017), and coefficients of the IPCC (2003; 2006).

Data analysis

The statistics of the IGN (2011-2015) were collected to calculate the volume by hectare of deadwood by hectare in forested areas of the seven French regions that make up the Rhône basin. The volume was divided by the area for spatialization.

Statistics spatialization

The merged forested areas from the IGN (2006-2017) were extracted to a point shape file of one hectare point distance, the same as the Rhône basin grid system, using *Join attributes by location* in QGIS and keeping all records.

The volume by hectare per region was allocated to their corresponding spatial units using a conditional statement in QGIS *Field Calculator* (**Appendix J¹**) and converted to 1 ha raster in SAGA-GIS *Shapes to Grid* using the Rhône basin grid system.

The tonnes of carbon per hectare (tC ha^{-1}) were calculated in SAGA-GIS *Grid Calculator* dividing the volume per area by four as in the previous section using the same coefficients for wood density and carbon content (**Figure 3.3**). I assumed the same amount of deadwood for the three reference years.

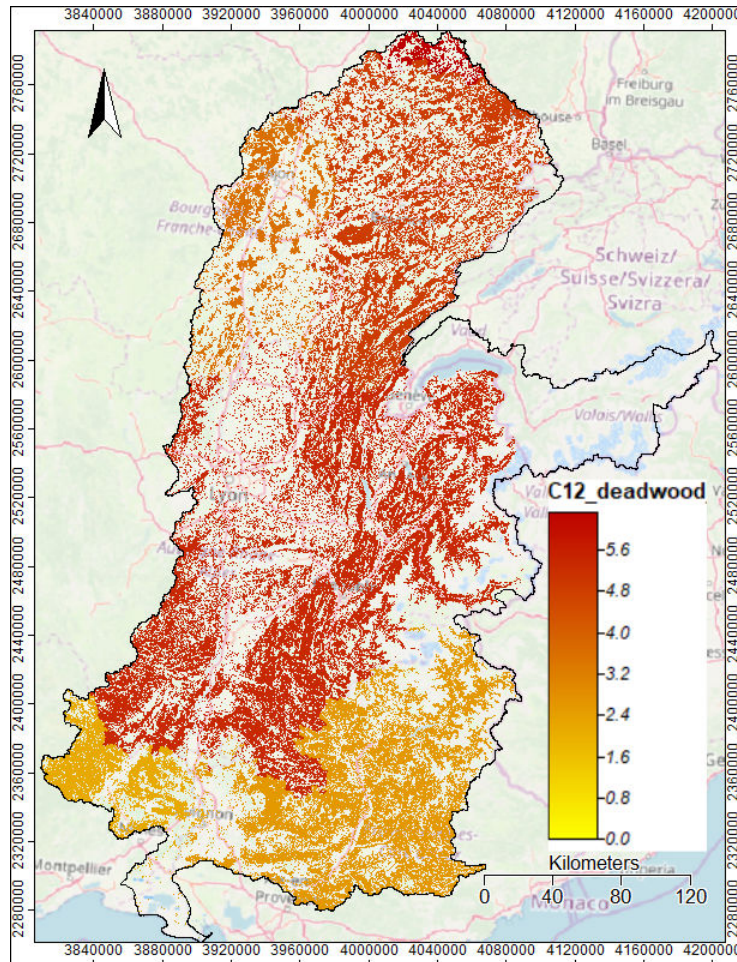


Figure 3.3. Bio-carbon in litter and deadwood ($tC\ ha^{-1}$). Spatialization of data from the *Inventaire Forestière National* at French department level, using QGIS and SAGA-GIS geoprocessing. Cumulative and integrated information by department reported in different years between 2011 and 2015.

Bio-carbon in soil

The bio-carbon in soil is a component of the opening stock. Two different approaches were explored: The first estimates organic carbon (Weber JL, personal communication) from the datasets of the European Soil Database and soil properties (ESDAC) based on the bulk density, the percentages of stones (Topsoil physical properties for Europe, 2015), and of organic carbon (Top Soil Organic Carbon, 2013). The second are estimations based on the National distribution of soil organic carbon in France (INRA, pedologic data convention, November 6, 2018).

Data analysis

For the first approach, the Top Soil Organic Carbon raster data were resampled to 500 m pixel size and multiplied by the bulk density converted to weight by area. The result was multiplied by the depth (30 cm), by the proportion of stones and organic carbon:

$Top\ Carbon\ (g\ cm^{-2}) = (Bulk\ Density \times 30\ cm) \times (stones\ percentage \div 100) \times (organic\ carbon\ percentage \div 100)$

The result was resampled to 1-ha pixel-size, and multiplied by 100 to convert the units to tonnes by hectare. To adjust for artificial, mineral, and water surfaces, the result was multiplied with SAGA-GIS *Calculator* by a reclassified dataset of land cover year 2000 (CORINE, 2017) and by the water surface. The reclassification of land cover was: artificial land as zero, except the discontinuous urban fabric as 0.5, the other categories as one. The initial dataset on water surface was converted to a 1-ha pixel-size raster (Hydrologie surfacique, 2006) (**Figure 3.4a**).

The second approach on the national distribution of soil organic carbon in France is an estimation using data mining of soil samples with a theoretic sampling distance of 16 km, and measurements *in situ* from 2000 to 2009, and a cross-validation scheme (Mulder et al., 2016). The 90-m-resolution product from INRA (2018) was resampled to 1-ha pixel-size in SAGA-GIS *Resampling*, and multiplied by a Rhône area mask to keep only the values corresponding to the watershed, and by 10 to convert $kgC\ m^{-2}$ to $tC\ ha^{-1}$ (**Figure 3.4b**).

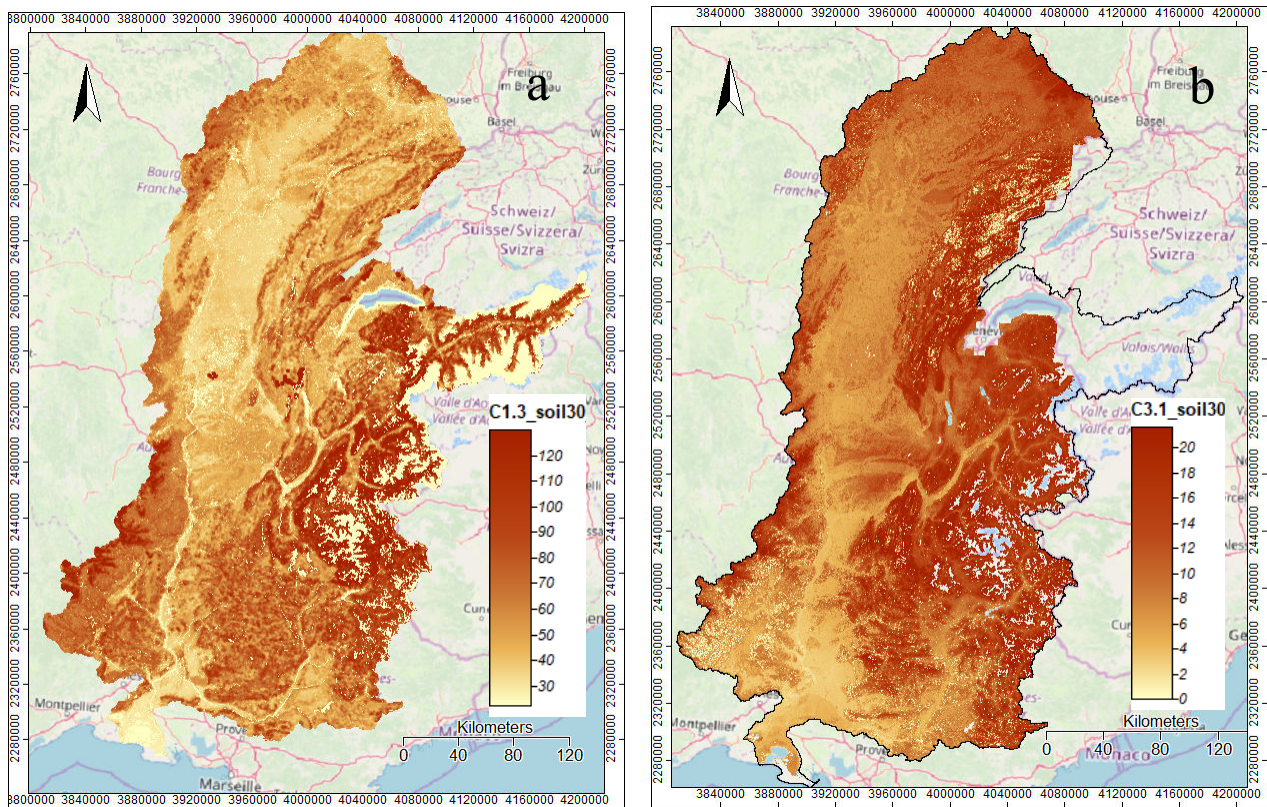


Figure 3.4. Bio-carbon in soil ($tC\ ha^{-1}$) for 0-30 cm layer. ESDAC estimations (2013-2015) (a) and INRA estimations (2000-2009) (b).

Chapter 3 Ecosystem Carbon Account

To estimate the values for the three dates, the INRA data resampled to 1-ha pixel-size was considered as the C stock in 2005, the percentage of organic carbon was multiplied by the geometric mean of the soil loss from 2001 and 2012 (ESDAC, 2019) to obtain an average of carbon loss per year. The stock per year (**Figure 3.5**) was calculated with the formulas:

$$C \text{ stock } 2000 = C \text{ stock } 2005 + 5 C \text{ loss mean}$$

$$C \text{ stock } 2006 = C \text{ stock } 2005 - C \text{ loss mean}$$

$$C \text{ stock } 2012 = C \text{ stock } 2005 - 7 C \text{ loss mean}$$

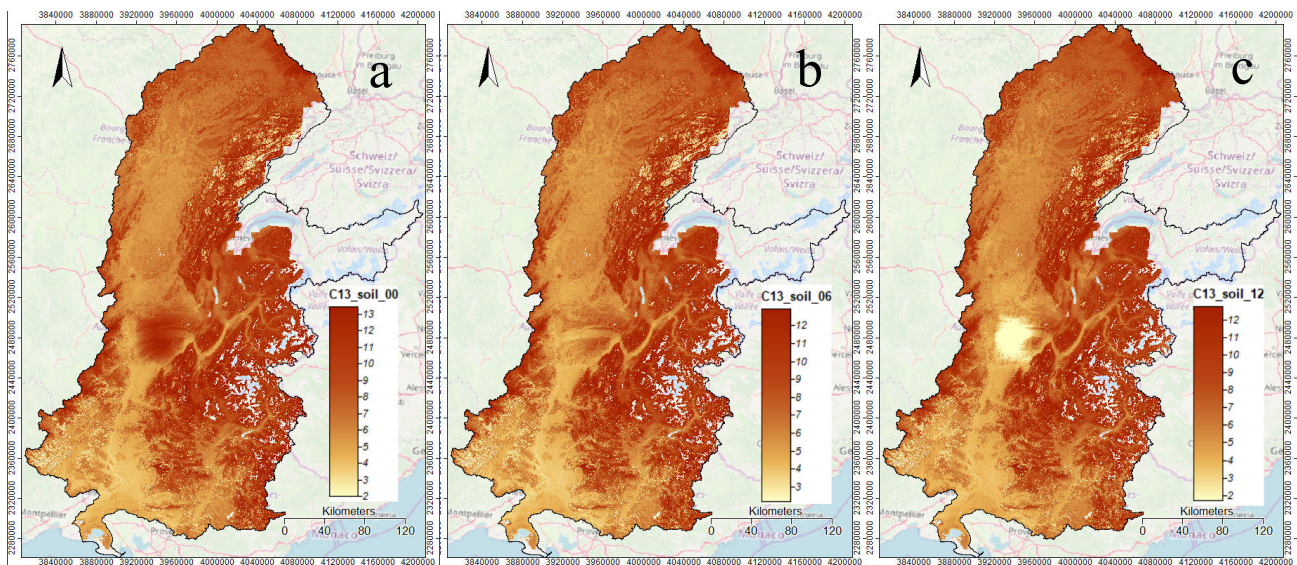


Figure 3.5. Estimations of bio-carbon in soil tC ha^{-1} for 0-30 cm layer, 2000 (a), 2006 (b), 2012 (c) calculated based on INRA dataset and soil loss from the ESDAC, using QGIS and SAGA-GIS geo-processing.

3.2.2 Flows

Gross Primary Production (GPP)

The GPP is the total energy assimilated by photosynthesis (Ricklefs and Miller, 1999). The GPP modelled by NASA-GPP (2000-2014) was used.

Spatial Data Assimilation

The weekly data available from NASA for 2000-2014 were downloaded (815 files) and the annual sums were calculated and artefactual (named no valid) values eliminated (**Appendix Jⁱ**). The results were multiplied by a scale factor of 0.0001 to obtain the values in kgC m^{-2} . They were reprojected to the ETRS89 Lambert Azimuthal Equal-Area projection using warp (reproject) Batch mode in QGIS. The QGIS point sampling tool was used to extract the values by year to a point file and rasterize them using the Rhône grid system in SAGA-GIS *Shapes to Grid*. The units were converted to $\text{kgC m}^{-2} = 10 \text{ tC ha}^{-1}$ in SAGA-GIS *Grid Calculator*. The file was clipped to obtain only the

French part of the watershed using a Rhône French departments mask and QGIS *Clip raster by mask layer* (**Figure 3.6**).

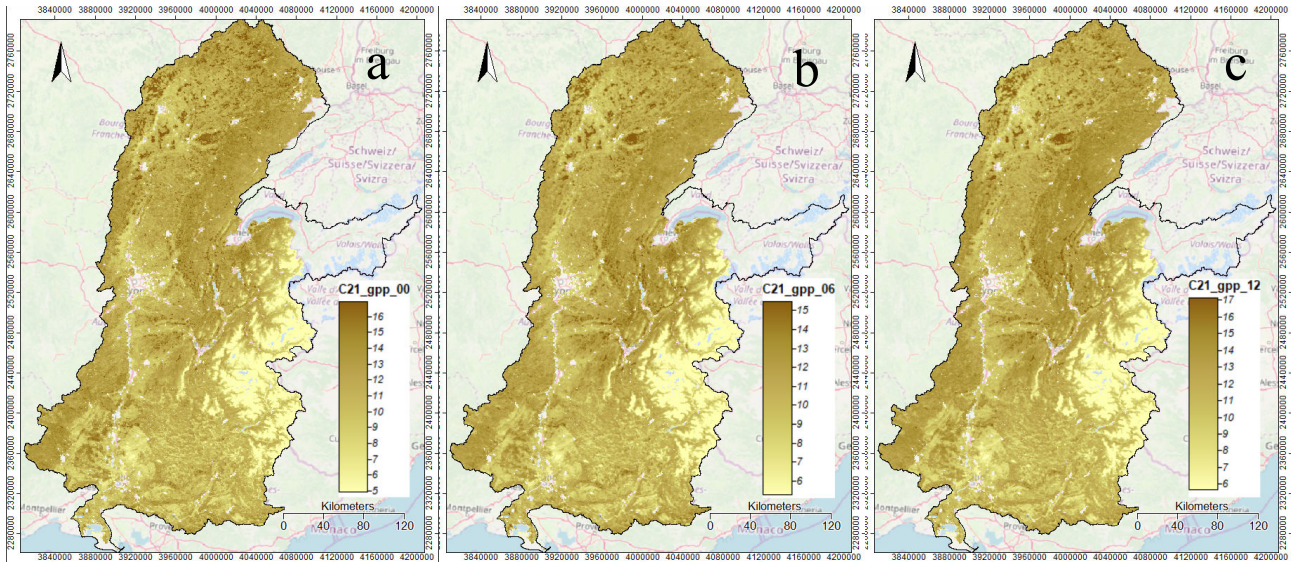


Figure 3.6. Gross Primary Production (GPP, tC ha^{-1}) 2000 (a), 2006 (b), and 2012 (c) annual sums using Python, QGIS and SAGA-GIS geo-processing tools.

Net Primary Production (NPP)

The NPP is the energy accumulated in plant biomass, plant growth and reproduction (Ricklefs and Miller, 1999). Two annual datasets were available: DLR-NPP (2003-2012) and NASA-NPP (2000-2014).

Spatial Data Assimilation

For the DLR-NPP (2003-2012), the time series was reprojected to the same projection as for GPP using warp (reproject) QGIS batch process. The QGIS point sampling tool was used to extract the values to a point file and rasterize them using the Rhône grid system in SAGA-GIS *Shapes to Grid*. The units were converted to $\text{tC ha}^{-1} = \text{kgC m}^{-2}/100$ in QGIS *Raster calculator*. The file was clipped to obtain only the French part of the watershed using a Rhône French departments mask and QGIS *Clip raster by mask layer* (**Figure 3.7**).

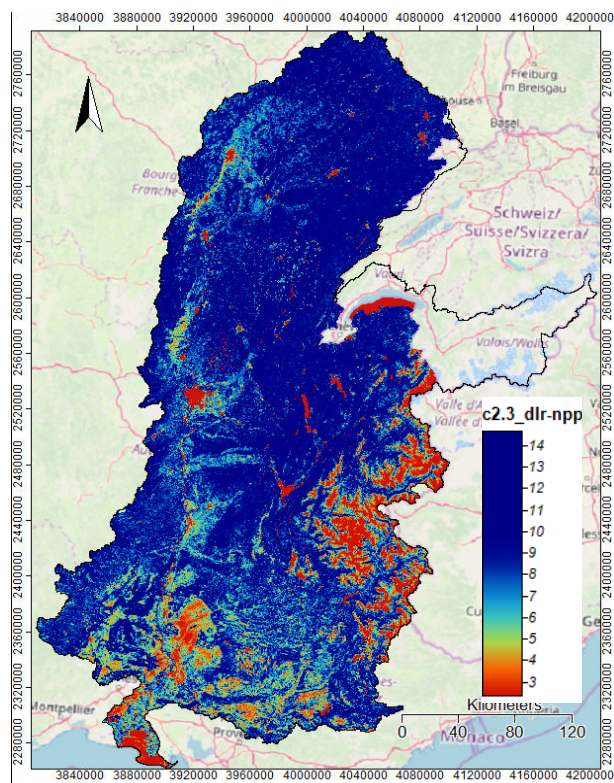


Figure 3.7. Net Primary Production (tC ha^{-1}) DLR estimations for 2006.

For years 2000, 2006 and 2012 of the NASA datasets, the no valid values (below to -3.27 kgC m^{-2}) and the projection ETRS89 Lambert Azimuthal Equal-Area were set and the original hdf4 file saved as tiff format in QGIS. The result was resampled to 1-ha pixel-size using the default parameters of SAGA-GIS *Resampling* and the Rhône grid system. Finally, the results were converted to $\text{kgC m}^{-2} = 10 \text{ tC ha}^{-1}$ in QGIS *Raster Calculator*.

As there was an inconsistency in the NASA time series for the first three years, the values for 2000 were estimated as 60% of the GPP according to the proportion calculated for the Rhône basin for 2004 to 2014. The global pattern calculated using the same data source is 52% (Zhang et al., 2009) (**Figure 3.8**).

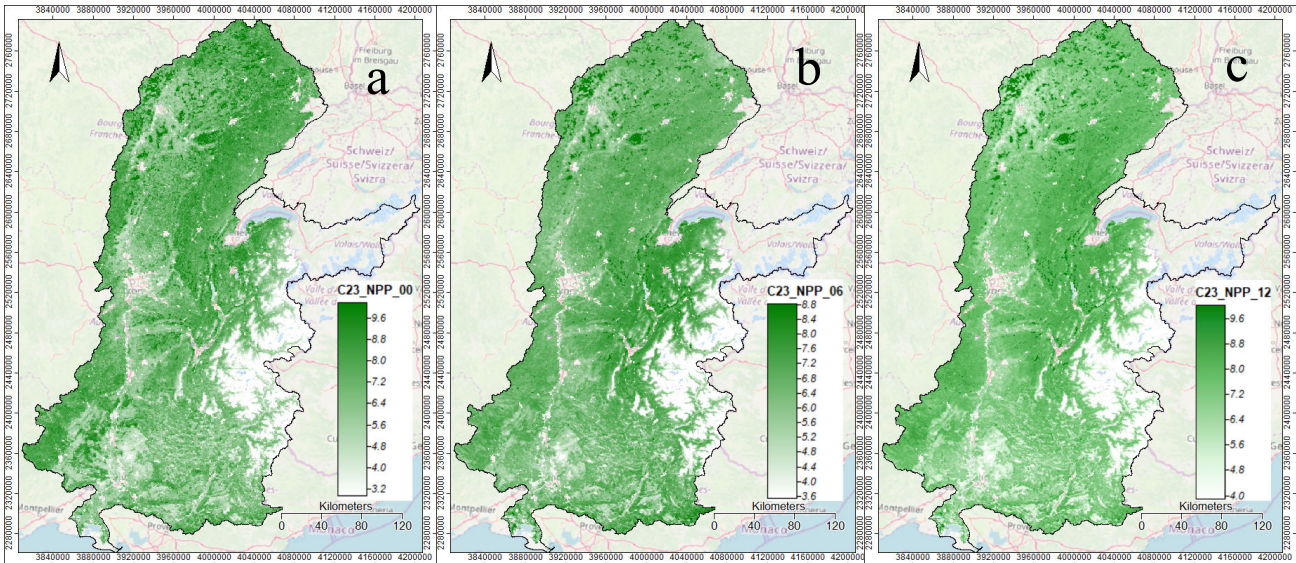


Figure 3.8. Net Primary Production (tC ha^{-1}) NASA estimations for 2000 (a), 2006 (b), and 2012 (c).

Autotrophic respiration (R_A)

The R_A is the respiration by photosynthetic organisms that represent energy lost (Ricklefs and Miller, 1999). It was calculated as $R_A = \text{NASA-GPP} - \text{NASA-NPP}$ for the years 2006 and 2012, and $R_A = \text{NASA-GPP} \times 0.4$ for the year 2000 due to the inconsistency in the original dataset.

Spatial Data Assimilation

The NASA-NPP kgC m^{-2} were either subtracted from the NASA-GPP annual sums of the previous section or multiplied with the QGIS *Raster Calculator*. The projection was set as ETRS89 Lambert Azimuthal Equal-Area saving the file as tiff format in QGIS. The result was resampled to 1-ha pixel-size using the default parameters of SAGA-GIS *Resampling* and the Rhône grid system. The units were converted using QGIS *Raster Calculator* as $\text{kgC m}^{-2} = 10 \text{ tC ha}^{-1}$. The files were clipped to obtain only the French part of the watershed using a departments mask and QGIS *Clip raster by mask layer* (Figure 3.9).

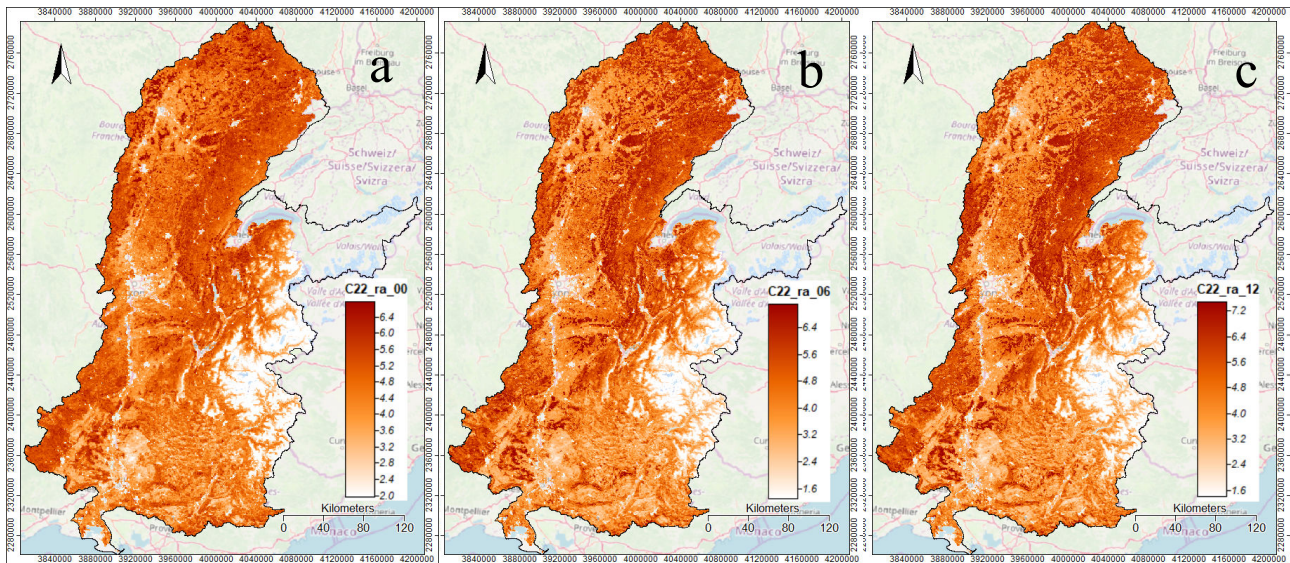


Figure 3.9. Autotrophic respiration (R_A , $tC\ ha^{-1}$) 2000 (a), 2006 (b), and 2012 (c). This is calculated as the difference between NASA GPP (Figure 3.6) and NPP (Figure 3.8) estimations using QGIS and SAGA-GIS geoprocessing tools. The range is from 0 (white) to around 7 (darkest color).

Heterotrophic respiration (R_H)

The R_H is a component of the bio-carbon inflow and corresponds to the respiration of organisms which obtain their energy and most of their nutrients by eating plants, animals or their dead remains (Ricklefs and Miller, 1999).

As only two points in the Rhône region were found in the Global Database of Soil Respiration Data (1961-2012), and two Eddy Fluxes sites in the European Fluxes Database Cluster (2019), I used the data of autotrophic respiration assuming that heterotrophic respiration equals autotrophic respiration in soil according to Kuzyakov and Larionova (2005), Olson et al. (2005), Tang and Baldocchi (2005), and Saurette et al. (2008). The two values of the Global Data Base were three times lower than the values assumed in the sampling sites (see Results section).

Net Ecosystem Production

The indicator is an important marker synthesizing the efficiency with which underlying processes operate and could bear the impacts of human activities.

This is calculated as the difference between NPP and R_H in the accounting tables. For the spatial representation, the R_A values before projection setting were subtracted from the pretreated (valid) NPP values in QGIS *Raster calculator*. The projection was set as ETRS89 Lambert Azimuthal Equal-Area, the file being saved as tiff format in QGIS. The result was resampled to 1-ha pixel-size

using the default parameters of SAGA-GIS *Resampling* and the Rhône grid system. The units were converted using QGIS *Raster Calculator* as $\text{kgC m}^{-2} = 10 \text{ tC ha}^{-1}$. The file was clipped to obtain only the French part of the watershed using a Rhône French departments mask and QGIS *Clip raster by mask layer* (see Results section).

3.2.3 Use

Agriculture harvested crops

The parameter is a component of the bio-carbon outflow. It represents the extraction of carbon through agriculture. The geographic data of the crop plots was extracted from the Graphical Land Parcels (RPG, 2012). The statistics were downloaded from the AGRESTE (2012).

Spatial Data Assimilation

A correspondence table was created to match the records from the two sources. In the RPG (2012) there are 25 types of crops, 19 of them in agreement with the official statistics (AGRESTE, 2018) for certain dates. For the years 2000, 2006, and 2012 a total of 14 crop matched spatially and temporally.

For the agricultural polygons two columns were created with the code and name of the departments in QGIS. Then the files of the 26 departments were merged, reprojected to the ETRS89 Lambert Azimuthal Equal-Area projection, joined by location to a point layer of 100 m of distance between points. The crops in the point layer were split selecting by expression in QGIS *Attribute table*, then saving only the selected features.

Each crop was joined with a .csv file containing the code department (dep) and the yield (reported as quintals ha^{-1}). The points were converted to 1 ha raster files in SAGA-GIS *Shapes to Grid* using the Rhône basin grid system.

To estimate bio-carbon, the yield was multiplied by 0.1 to convert to tonnes, then by 0.2 or 0.8 whether the crop was wet or dry, respectively, to obtain the dry biomass (except for forage crops that were already reported as dry biomass). A factor of 0.5 was used to calculate carbon (IPCC, 2006). For forage crops the tonnes of C were divided by the corresponding surface in the given department.

The number of crops by year were merged in SAGA-GIS *Mosaicking* (**Figure 3.10**).

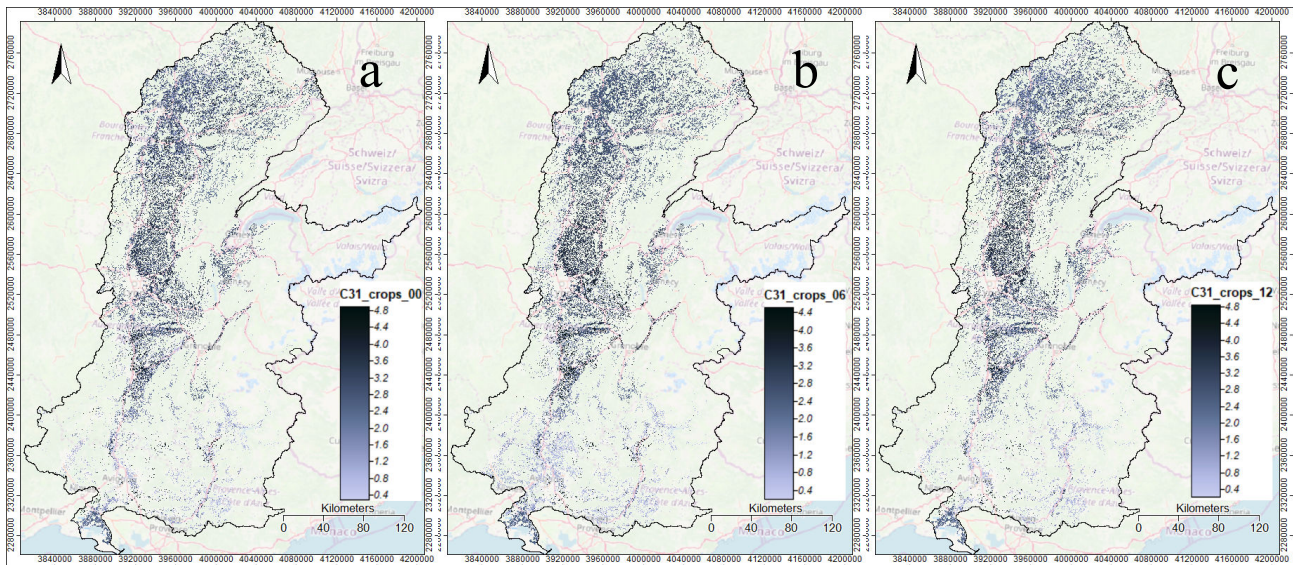


Figure 3.10. Agriculture harvested crops (tC ha⁻¹) 2000 (a), 2006 (b), and 2012 (c). Spatialization of data on harvested crops from the *Ministère de l'agriculture et de l'alimentation* at French department level and by type of crop, using QGIS and SAGA-GIS geo-processing tools. The range is from 0 (white) to around 5 (darkest color).

Wood removals

As component of the bio-carbon outflow, wood removals represent the extraction of carbon through forestry. The vector database of forested area from IGN (1992-2004) served for the spatialization of the wood removal statistics (except energy) for 2000 (source Weber November 15, 2018), 2006, and 2012 (AGRESTE, 2005-2017).

Data analysis

The forest area in hectares by department was calculated using QGIS GroupStats. The volume (m³) of wood removals was converted to tonnes of carbon using the same coefficients as for living trees and divided by the area of forest by department.

Spatial Data Assimilation

The results were joined to the forested area in QGIS and converted to raster files in SAGA-GIS *Shapes to Grid* using the Rhône basin grid system (**Figure 3.11**).

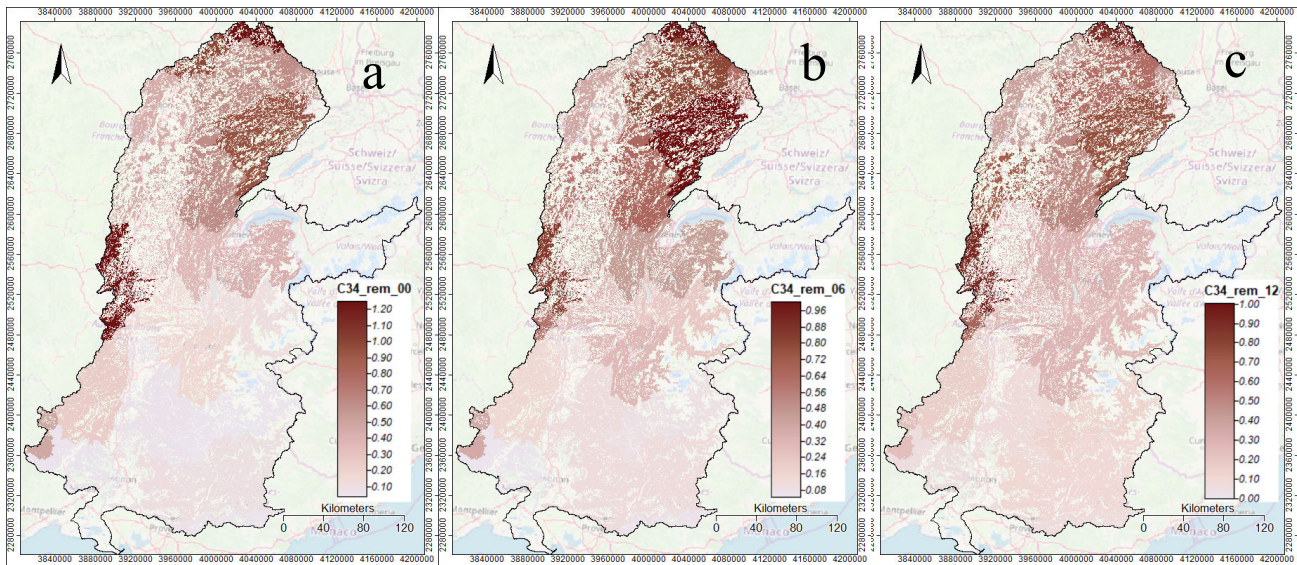


Figure 3.11. Wood removals (tC ha^{-1}) at department level in 2000 (a), 2006 (b), and 2012 (c). Spatialization of data on wood removals from the *Ministère de l'agriculture et de l'alimentation* at French department level, using QGIS and SAGA-GIS geo-processing tools. The range is from 0 (white) to 2.28 in a, 1.17 in b, and 0.90 in c (darkest color).

Withdrawals of animal bio-carbon

This is a component of the bio-carbon outflow. It represents the extraction of carbon through animal husbandry. The cattle statistics were downloaded from AGRESTE (2000-2015) and spatialized in the pasture class years 2000, 2006, and 2012 from CORINE (2017).

Spatial Data Assimilation

The pasture areas (class 18 from the raster data CORINE, 2017) were obtained truncating this class from the original files using SAGA-GIS *Calculator* and a conditional statement to change the code of the class as one and other classes as zero. The area was calculated adding up the number of cells (1 ha) with *Grid Statistics for Polygons* in SAGA-GIS using a department vector layer. The number of heads by department (AGRESTE 2000-2015) were joined in QGIS, divided by the pasture area, and converted to tonnes by hectare in QGIS *field calculator* multiplying by the cattle mean weight of 0.5 tonnes (FAO, 1991) and using a coefficient of carbon content in animals of 0.1. Thus:

$$\text{Bio-carbon of cattle} = \text{Head of cattle by hectare} \times \text{mean weight} \times \text{coefficient carbon content}$$

The result was converted to raster files in SAGA-GIS *Shapes to Grid* using the Rhône basin grid system, multiplied by the pasture areas. The file was clipped to obtain only the French part of the watershed using a Rhône French departments mask and QGIS *Clip raster by mask layer*. (**Figure 3.12**).

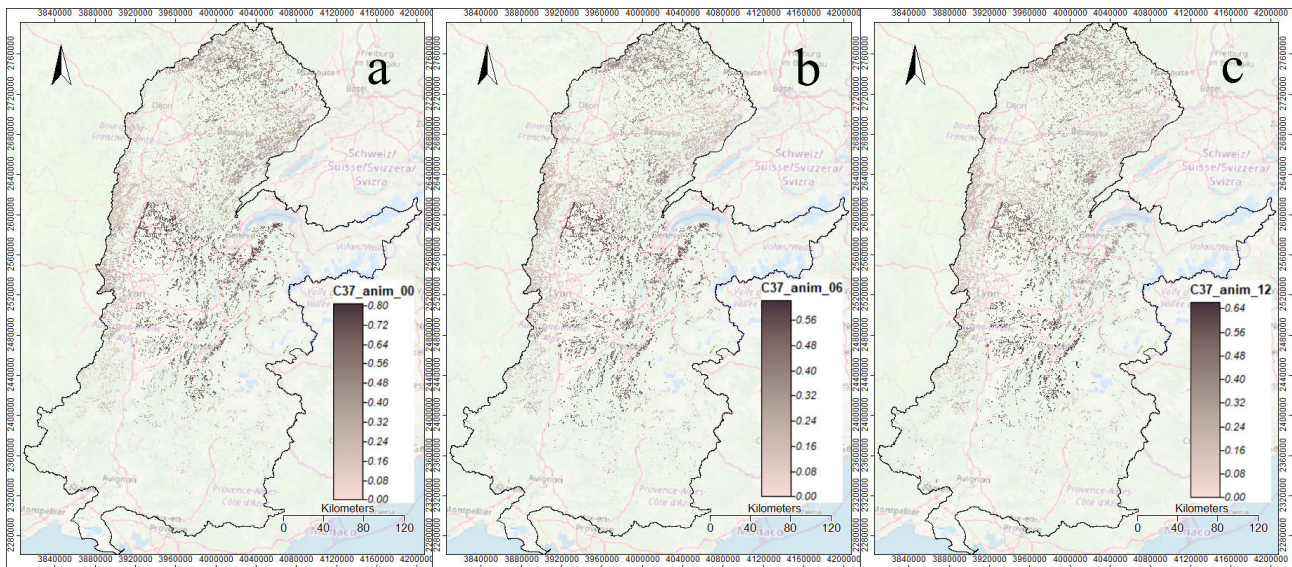


Figure 3.12. Withdrawals of animal bio-carbon (cattle, tC ha⁻¹) at department level 2000 (a), 2006 (b), and 2012 (c). Spatialization of data on cattle from the *Ministère de l'agriculture et de l'alimentation* at French department level, using QGIS and SAGA-GIS geo-processing tools. The range is from 0 (white) to 1.43 in *a*, and 1.26 in *b* and *c* (darkest color).

Loss of bio-carbon due to land use change

To estimate the loss of carbon due to land use, the number of hectares and the type of change of land use were computed from the land cover CORINE (2017) dataset for changes from 2000 to 2006 in the Rhône watershed (see Land Cover, Chapter 2). The sum of the corresponding type stocks was multiplied by the number of hectares changed. This period was assumed as the median and used for the other two periods.

3.2.4 Storing and retrieving spatial data

The datasets of the variables studied were sent to the *eccounts_rho* PostgreSQL database with the geospatial extension PostGIS in the *Pôle Scientifique de Modélisation Numérique (PSMN)* server (**Appendix L**ⁱ). Geographic spatial queries in PostgreSQL were performed to add up the pixels by each carbon item and Dominant Land Cover Type (DLCT; **Appendix M**ⁱ) to fill the carbon balance sheet. The queries results were validated using *Raster layer statistics* in QGIS, and summing up the values of the pixels by DLCT polygon in *Grid Statistics for Polygons* in SAGA-QGIS and by DLCT class in *Group Stats* QGIS.

The DLCTs represent the probability surfaces (varying from 0 to 100) for the presence of a certain land cover class within a smoothing radius. A Gaussian function was used to weight this

information according to the distance from a considered point in kilometres to get the “intensities or potential in the neighbourhood” (Páramo, 2008). The 44 classes from CORINE (2017) 250 m pixel-size were aggregated in 10 classes, each class was smoothed with SAGA-GIS Gaussian Filter (radius of 2.5 km) and selected as dominant for those over 25% (**Appendix E**¹). The classes were reclassified and merged in SAGA-GIS *Mosaicking*, arranging them according to their size to minimize the relative error (those with the larger areas at the bottom and those with the smaller areas at the top). The pixels with less than 25% were classified as “not dominant with human activities” and “not dominant natural” according to CORINE classification. These two classes were sieved out in QGIS for zones smaller than 400 ha (**Figure 3.13**).

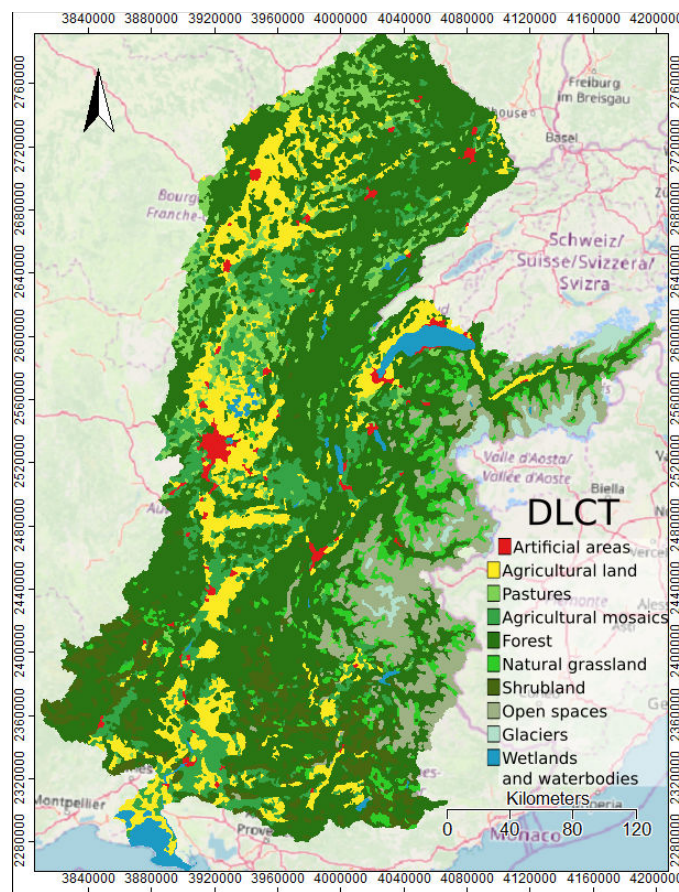


Figure 3.13. Dominant Land Cover Types (DLCT) from the aggregation of the 44 CLC classes in ten and their potential in the neighborhood. QGIS and SAGA-GIS geo-processing tools were used.

The change in percentage was calculated by Socio-Ecological Landscape Unit (SELU; see Chapter 5, section 4.2.3) with the formula:

$$\% \text{ increase} = \text{increase} \div \text{original amount} \times 100$$

$$\text{increase} = \text{final amount} - \text{original amount}$$

3.3 Results Carbon balance

A coherent carbon balance was established for the years 2000, 2006, and 2012 on the French part of the Rhône watershed according to the CORINE (2017) data series. The accounts were calculated using 21 different data sources from a total of 48 consulted. They cover 25% of the total carbon accounting items in the balance sheets. The columns of the table show the total amount of carbon by Dominant Land Cover Type and the total for the Rhône watershed (**Table 3.1**). This is a compact version of the genuine accounting tables from the ENCA-QSP. These accounting tables generate a detailed description of the categories represented in the work diagram (**Figure 3.1**). They describe the ecosystem carbon basic balance and define the Net Ecosystem Carbon Balance. Next, the Net Ecosystem Accessible Carbon Surplus is calculated, followed by total use. The whole allows calculating indexes of the intensity of use.

Table 3.1. Carbon balance sheet 2000 (a), 2006 (b), and 2012 (c) compact versions.

Ecosystem Carbon Account 2000												
a	SEEA-EEA & ENCA-QSP land cover ecosystem units										TOTAL tC	
	DLCT 1	DLCT 2	DLCT 3	DLCT 4	DLCT 5	DLCT 6	DLCT 7	DLCT 8	DLCT 9	DLCT 10		
	Artificial	Agricultural land	Pastures	Agricultural mosaics	Forest	Natural grassland	Shrubland and heathland	Open spaces	Glaciers and perpetual snow	Wetlands and water bodies		
Area (ha)	136381	1049546	549219	968854	4474305	395391	543494	646937	55395	147190		
IPCC land use classification	SL = Settlements	CL = Cropland			GL = Grassland	FL = Forest Land	OL = Other Land			WL = Wetlands		
I. Ecosystem Carbon Basic Balance												
C1	Opening Stocks	2394968.93	19306551.73	12708491.72	23082403.45	217906007.36	15866385.97	17336726.85	14514435.05	128467.41	2344446.80	325,588,885.26
C2.a	NEP (Net Ecosystem Production) = C2.3-C2.4	49913.25	1874303.87	1190889.82	1868262.85	11403294.39	605947.82	1086632.00	532978.13	2871.95	132032.50	18,747,126.58
C2	Total inflow of biocarbon (gains) = C2.a+C2.b	49913.25	1874303.87	1190889.82	1868262.85	11403294.39	605947.82	1086632.00	532978.13	2871.95	132032.50	18,747,126.58
C3.a	Harvest of agriculture crops, wood & other vegetation	40119.66	1255708.70	406051.71	771510.78	2561660.07	40270.81	74251.70	17489.67	23.62	36672.87	5,203,759.60
C3.b	Withdrawals of secondary biocarbon	1560.06	19081.98	113466.95	39396.96	238509.35	2478.96	1697.58	534.24	0.00	1479.24	
C3	Total withdrawals of biocarbon = C3.a+C3.b	41679.72	1274790.68	519518.67	810907.74	2800169.42	42749.77	75949.28	18023.91	23.62	38152.11	5,621,964.92
C4	Net indirect anthropogenic losses of biocarbon & biofuel combustion	86052.62	1798.03	0	0	0	0	0	0	0	0	
C5	Total use of ecosystem biocarbon = C3+C4	127732.34	1276588.70	519518.67	810907.74	2800169.42	42749.77	75949.28	18023.91	23.62	38152.11	5,709,815.56
C6	Natural processes and disturbances											
C7	Total outflow of biocarbon (losses)	127732.34	1276588.70	519518.67	810907.74	2800169.42	42749.77	75949.28	18023.91	23.62	38152.11	5,709,815.56
C8.1	Net Ecosystem Carbon Balance 1, NECB 1 [Flows] = Inflows - Outflows = C2-C7	-77,819.09	597,715.17	671,371.15	1,057,355.12	8,603,124.97	563,198.05	1,010,682.72	514,954.22	2,848.33	93,880.39	13,037,311.02
C8.2	Adjustment and reappraisals											
C8.3	NECB 2 [Stocks] = Change of biocarbon stocks											
C9	Closing Stocks = C1+C8.1+C8.2 or = C2+C8.3	2317149.84	19904266.90	13379862.87	24139758.56	226509132.32	16429584.02	18347409.56	15029389.27	131315.74	2438327.19	338,626,196.28
II. Accessible Resource Surplus												
C2	Total inflow of biocarbon (gains) = C2.a+C2.b	49913.25	1874303.87	1190889.82	1868262.85	11403294.39	605947.82	1086632.00	532978.13	2871.95	132032.50	18,747,126.58
C10	Accessibility net correction											
C11	Net Ecosystem Accessible Carbon Surplus = C2 + C10	49913.25	1874303.87	1190889.82	1868262.85	11403294.39	605947.82	1086632.00	532978.13	2871.95	132032.50	18,747,126.58
III. Total Uses of Ecosystem Bio and Geo-Carbon												
C5	Total use of ecosystem biocarbon = C3+C4	127732.34	1276588.70	519518.67	810907.74	2800169.42	42749.77	75949.28	18023.91	23.62	38152.11	5,709,815.56
IV. Table of indexes of intensity of use and ecosystem health												
SCU	Sustainable intensity of carbon use = C11/C5	0.39	1.47	2.29	2.30	4.07	14.17	14.31	29.57	121.57	3.46	
CEH	Composite ecosystem biocarbon health index											
CIP [IUUV]	Biocarbon ecological internal unit value = AVG(SCU+CEH)	0.39	1.47	2.29	2.30	4.07	14.17	14.31	29.57	121.57	3.46	

Ecosystem Carbon Account 2006												
b	SEEA-EEA & ENCA-QSP land cover ecosystem units										TOTAL tC	
	DLCT 1	DLCT 2	DLCT 3	DLCT 4	DLCT 5	DLCT 6	DLCT 7	DLCT 8	DLCT 9	DLCT 10		
	Artificial	Agricultural land	Pastures	Agricultural mosaics	Forest	Natural grassland	Shrubland and heathland	Open spaces	Glaciers and perpetual snow	Wetlands and water bodies		
Area (ha)	136381	1049546	549219	968854	4474305	395391	543494	646937	55395	147190		
IPCC land use classification	SL = Settlements	CL = Cropland			GL = Grassland	FL = Forest Land	OL = Other Land			WL = Wetlands		
I. Ecosystem Carbon Basic Balance												
C1	Opening Stocks	2464987.67	19818853.99	12913910.75	23792695.45	217636567.32	16420077.67	17627471.53	15581179.87	200829.64	2542667.81	328,999,241.70
C2.a	NEP (Net Ecosystem Production) = C2.3-C2.4	60180.85	2296944.06	946479.08	2087142.54	8576058.98	819974.33	1184511.05	958886.50	8212.56	151998.85	17,090,388.81
C2	Total inflow of biocarbon (gains) = C2.a+C2.b	60180.85	2296944.06	946479.08	2087142.54	8576058.98	819974.33	1184511.05	958886.50	8212.56	151998.85	17,090,388.81
C3.a	Harvest of agriculture crops, wood & other vegetation	24807.26	1184068.45	379880.20	734557.16	2496995.11	52331.45	69879.82	25684.91	25.09	37713.47	
C3.b	Withdrawals of secondary biocarbon	932.58	15037.74	87277.14	30287.88	185363.81	2060.82	1876.68	428.40	0.00	980.46	
C3	Total withdrawals of biocarbon = C3.a+C3.b	25739.84	1199106.19	467157.34	764845.03	2682358.92	54392.27	71756.50	26113.31	25.09	38693.93	5,330,188.42
C4	Net indirect anthropogenic losses of biocarbon & biofuel combustion	86052.615	1798.025	0	0	0	0	0	0	0	0	
C5	Total use of ecosystem biocarbon = C3+C4	111792.45	1200904.22	467157.34	764845.03	2682358.92	54392.27	71756.50	26113.31	25.09	38693.93	5,418,039.06
C6	Natural processes and disturbances											
C7	Total outflow of biocarbon (losses)	111792.45	1200904.22	467157.34	764845.03	2682358.92	54392.27	71756.50	26113.31	25.09	38693.93	
C8.1	Net Ecosystem Carbon Balance 1, NECB 1 [Flows] = Inflows - Outflows = C2-C7	-51,611.61	1,096,039.84	479,321.74	1,322,297.51	5,893,700.07	765,582.06	1,112,754.55	932,773.19	8,187.47	113,304.92	
C8.2	Adjustment and reappraisals											
C8.3	NECB 2 [Stocks] = Change of biocarbon stocks											
C9	Closing Stocks = C1+C8.1+C8.2 or = C1+C8.3	2413376.07	20914893.83	13393232.49	25114992.96	223530267.39	17185659.73	18740226.08	16513953.06	209017.11	2655972.73	340,671,591.45
II. Accessible Resource Surplus												
C2	Total inflow of biocarbon (gains) = C2.a+C2.b	60180.85	2296944.06	946479.08	2087142.54	8576058.98	819974.33	1184511.05	958886.50	8212.56	151998.85	17,090,388.81
C10	Accessibility net correction											
C11	Net Ecosystem Accessible Carbon Surplus = C2 + C10	60180.85	2296944.06	946479.08	2087142.54	8576058.98	819974.33	1184511.05	958886.50	8212.56	151998.85	17,090,388.81
III. Total Uses of Ecosystem Bio and Geo-Carbon												
C5	Total use of ecosystem biocarbon = C3+C4	111792.45	1200904.22	467157.34	764845.03	2682358.92	54392.27	71756.50	26113.31	25.09	38693.93	5,418,039.06
IV. Table of indexes of intensity of use and ecosystem health												
SCU	Sustainable intensity of carbon use = C11/C5	0.54	1.91	2.03	2.73	3.20	15.08	16.51	36.72	327.29	3.93	
CEH	Composite ecosystem biocarbon health index											
CIP [IUUV]	Biocarbon ecological internal unit value = AVG(SCU+CEH)	0.54	1.91	2.03	2.73	3.20	15.08	16.51	36.72	327.29	3.93	

Ecosystem Carbon Account 2012												
C	SEEA-EEA & ENCA-QSP land cover ecosystem units										TOTAL tC	
	DLCT 1	DLCT 2	DLCT 3	DLCT 4	DLCT 5	DLCT 6	DLCT 7	DLCT 8	DLCT 9	DLCT 10		
	Artificial	Agricultural land	Pastures	Agricultural mosaics	Forest	Natural grassland	Shrubland and heathland	Open spaces	Glaciers and perpetual snow	Wetlands and water bodies		
Area (ha)	136381	1049546	549219	968854	4474305	395391	543494	646937	55395	147190		
IPCC land use classification	SL = Settlements	CL = Cropland			GL = Grassland	FL = Forest Land	OL = Other Land			WL = Wetlands		
I. Ecosystem Carbon Basic Balance												
C1	Opening Stocks	2501175.25	19878347.75	13125000.44	24047394.33	223181909.57	16565306.62	18949939.65	15543579.14	200224.34	2576005.53	336,568,882.63
C2.a	NEP (Net Ecosystem Production) = C2.3-C2.4	71504.07	2703283.11	1247444.12	2503649.24	11479176.73	932357.03	1389468.58	1083693.40	9114.73	178544.76	21,598,235.77
C2	Total inflow of biocarbon (gains) = C2.a+C2.b	71504.07	2703283.11	1247444.12	2503649.24	11479176.73	932357.03	1389468.58	1083693.40	9114.73	178544.76	21,598,235.77
C3.a	Harvest of agriculture crops, wood & other vegetation	24102.79	1187274.33	381764.46	745502.65	2265864.59	48294.56	65335.96	26186.88	17.62	36382.08	
C3.b	Withdrawals of secondary biocarbon	877.14	15557.58	86359.50	31300.20	188021.33	2090.70	1934.64	472.32	0.00	991.26	
C3	Total withdrawals of biocarbon = C3.a+C3.b	24979.93	1202831.91	468123.96	776802.85	2453885.92	50385.26	67270.60	26659.20	17.62	37373.34	5,108,330.57
C4	Net indirect anthropogenic losses of biocarbon & biofuel combustion	86052.615	1798.025	0	0	0	0	0	0	0	0	
C5	Total use of ecosystem biocarbon = C3+C4	111032.54	1204629.93	468123.96	776802.85	2453885.92	50385.26	67270.60	26659.20	17.62	37373.34	5,196,181.21
C6	Natural processes and disturbances											
C7	Total outflow of biocarbon (losses)	111032.54	1204629.93	468123.96	776802.85	2453885.92	50385.26	67270.60	26659.20	17.62	37373.34	
C8.1	Net Ecosystem Carbon Balance 1, NECB 1 [Flows] = Inflows - Outflows = C2-C7	-39,528.47	1,498,653.17	779,320.16	1,726,846.39	9,025,290.81	881,971.77	1,322,197.99	1,057,034.21	9,097.11	141,171.42	
C8.2	Adjustment and reappraisals											
C8.3	NECB 2 [Stocks] = Change of biocarbon stocks											
C9	Closing Stocks = C1+C8.1+C8.2 or = C1+C8.3	2461646.78	21377000.93	13904320.60	25774240.71	232207200.38	17447278.39	20272137.64	16600613.35	209321.45	2717176.95	352,970,937.18
II. Accessible Resource Surplus												
C2	Total inflow of biocarbon (gains) = C2.a+C2.b	71504.07	2703283.11	1247444.12	2503649.24	11479176.73	932357.03	1389468.58	1083693.40	9114.73	178544.76	21,598,235.77
C10	Accessibility net correction											
C11	Net Ecosystem Accessible Carbon Surplus = C2 + C10	71504.07	2703283.11	1247444.12	2503649.24	11479176.73	932357.03	1389468.58	1083693.40	9114.73	178544.76	21,598,235.77
III. Total Uses of Ecosystem Bio and Geo-Carbon												
C5	Total use of ecosystem biocarbon = C3+C4	111032.54	1204629.93	468123.96	776802.85	2453885.92	50385.26	67270.60	26659.20	17.62	37373.34	5,196,181.21
IV. Table of indexes of intensity of use and ecosystem health												
SCU	Sustainable intensity of carbon use = C11/C5	0.64	2.24	2.66	3.22	4.68	18.50	20.65	40.65	517.39	4.78	
CEH	Composite ecosystem biocarbon health index											
CIP [IUW]	Biocarbon ecological internal unit value = AVG(SCU+CEH)	0.64	2.24	2.66	3.22	4.68	18.50	20.65	40.65	517.39	4.78	

3.3.1 Stocks

The aboveground living biomass regarding to trees has a mean value of 36.94 tC ha⁻¹ with SD of 18.26 tC ha⁻¹ in 2000, 38.73 tC ha⁻¹ with SD of 19.17 tC ha⁻¹ in 2006, and 40.31 tC ha⁻¹ with SD of 19.21 tC ha⁻¹ in 2012. These averages are similar to the estimated carbon amount for forest in France 39.5 32 tC ha⁻¹ (IFN, 2006-2010) and Europe 32 tC ha⁻¹ (FAO, 2001). The total stock of carbon in French forest trees, including roots (1,226,258,038 tC, IGN, 2018), is five times higher than the estimations for the Rhône watershed.

The deadwood average of 4.23 tC ha⁻¹ with SD of 1.25 tC ha⁻¹ is higher than the national average of 2.00 tC ha⁻¹. The total deadwood in France represents 5% of living trees (IFN, 2006-2010), which is below to the percentage calculated here (10%).

The soil average for the top 30 cm is 7.86 tC ha⁻¹ with SD 2.78 tC ha⁻¹ in 2000, 7.54 tC ha⁻¹ with SD 2.68 tC ha⁻¹ in 2006, and 7.22 tC ha⁻¹ with SD 2.77 tC ha⁻¹ in 2012. Highest values have been reported at high altitudes (INRA, 2017).

The combination of the three items of carbon stocks (**Figure 3.14**) gives an average value of 36.21 tC ha⁻¹ with SD of 23.09 tC ha⁻¹ in 2000, 37.21 tC ha⁻¹ with SD of 24.16 tC ha⁻¹ in 2006, and 38.06 tC ha⁻¹ with SD of 24.59 tC ha⁻¹ in 2012.

In summary, the results indicate that the carbon stocks are relatively stable over time in the watershed. The areas where the carbon stocks are diminished more pronouncely correspond to Agricultural land in the south of Lyon, to Forest in the northeast of the watershed, and to Open spaces in the Alps. Positive changes occurred in the south of the watershed corresponding to Forest and Shrubland (**Figure 3.14d**).

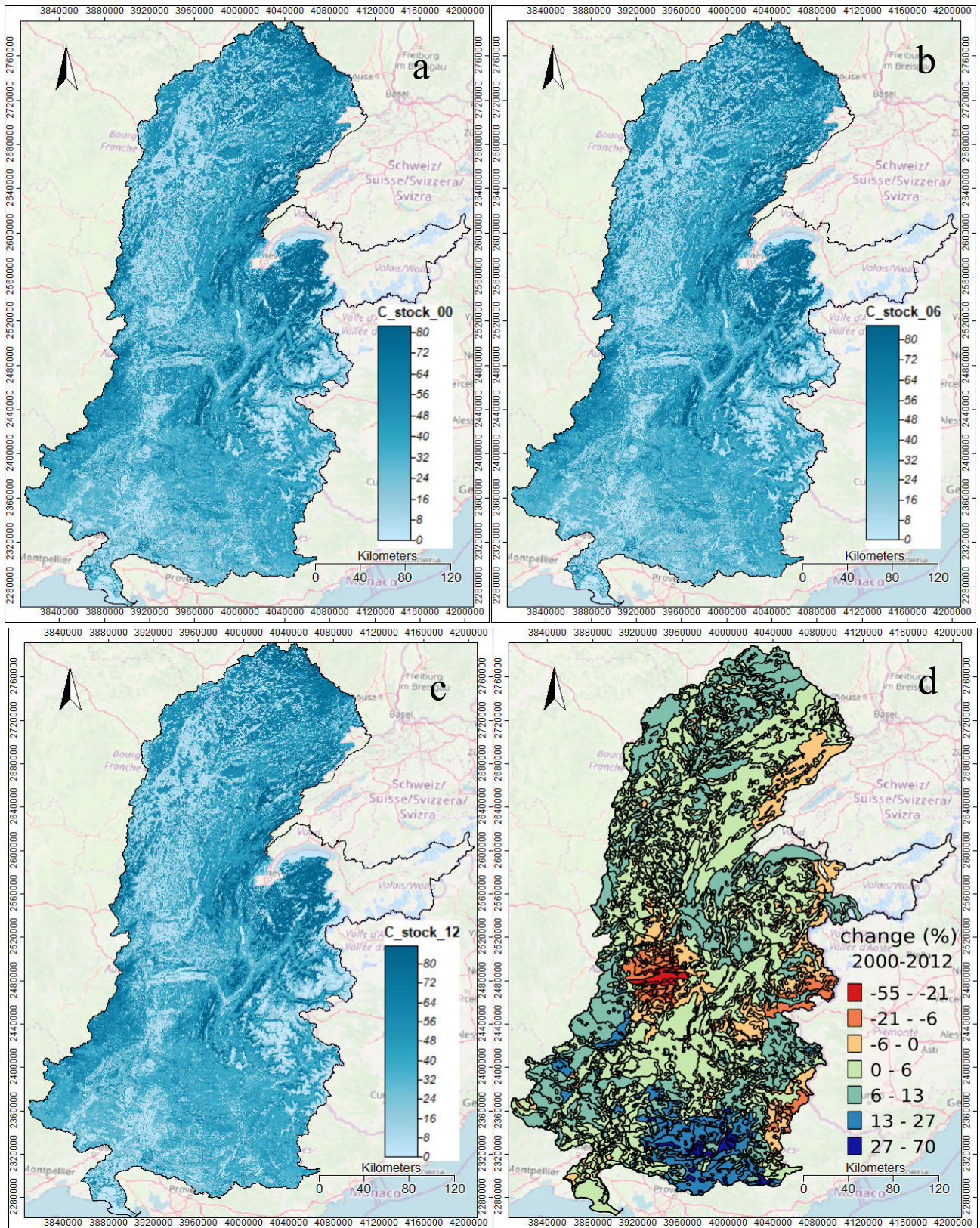


Figure 3.14. Carbon stocks (tC ha^{-1}) by integration of data on forest (Figure 3.2), deadwood (Figure 3.3), and soil (Figure 3.5) in 2000 (a), 2006 (b), 2012 (c). (d) shows the change (%) from 2000 to 2012, information in transboundary sub-basins relies only on data in the French part. The cool colors indicate an increase of stock. QGIS and SAGA-GIS geo-processing tools were used.

3.3.2 Flows

The GPP computed from NASA sources had an average of 10.91 tC ha⁻¹ with SD of 3.00 tC ha⁻¹ in 2000, 10.35 tC ha⁻¹ with SD of 2.55 tC ha⁻¹ in 2006, and 11.98 tC ha⁻¹ with SD of 2.87 tC ha⁻¹ in 2012. This model exceeds tower-based calculations by 20%–30% with an error estimated at 28% according to Heinsch et al. (2006).

The autotrophic respiration had an average of 4.36 tC ha⁻¹ with SD of 1.2 tC ha⁻¹ in 2000, 4.18 tC ha⁻¹ with SD of 1.45 tC ha⁻¹ in 2006, and 4.43 tC ha⁻¹ with SD of 1.52 tC ha⁻¹ in 2012.

The NASA-NPP in 2000 had a mean of 6.54 tC ha⁻¹ with SD of 1.81 tC ha⁻¹, in 2006 6.17 tC ha⁻¹ with SD of 1.32 tC ha⁻¹, and in 2012 6.95 tC ha⁻¹ with SD of 1.53 tC ha⁻¹. NASA values were around 20% lower than the DLR-NPP model. However, they are comparable to the maximum value estimated for Germany by the DLR (6.62 tC ha⁻¹; Tum and Günther, 2011). According to the authors, the DLR model is an overestimate of statistical yield data for Germany and Austria by 13% in 2000, and 1% to 2% in 2001.

In summary, the average values of carbon flows appear relatively stable over time.

The estimated Net Ecosystem Production (**Figure 3.15**) is on average 2.18 tC ha⁻¹ with SD of 0.60 tC ha⁻¹ in 2000, 1.99 tC ha⁻¹ with SD of 0.83 tC ha⁻¹ in 2006, and 2.51 tC ha⁻¹ with SD of 0.77 tC ha⁻¹ in 2012, the results are comparable to estimates based on Eddy Covariance (1.93 tC ha⁻¹) in beech forests in Denmark (Canadell et al., 2000). The reduction in the average production in 2006 seems to be related to high temperatures episodes in that year (Meteo France).

Concerning the spatialization pattern of the rate of the NEP change, the comparison to the corresponding distribution of Dominant Land Cover Types is shown in **Figure 3.15 d and e**.

The comparison suggests that agriculture and protected areas stand out in a contrasting manner with respect to NEP rates of change.

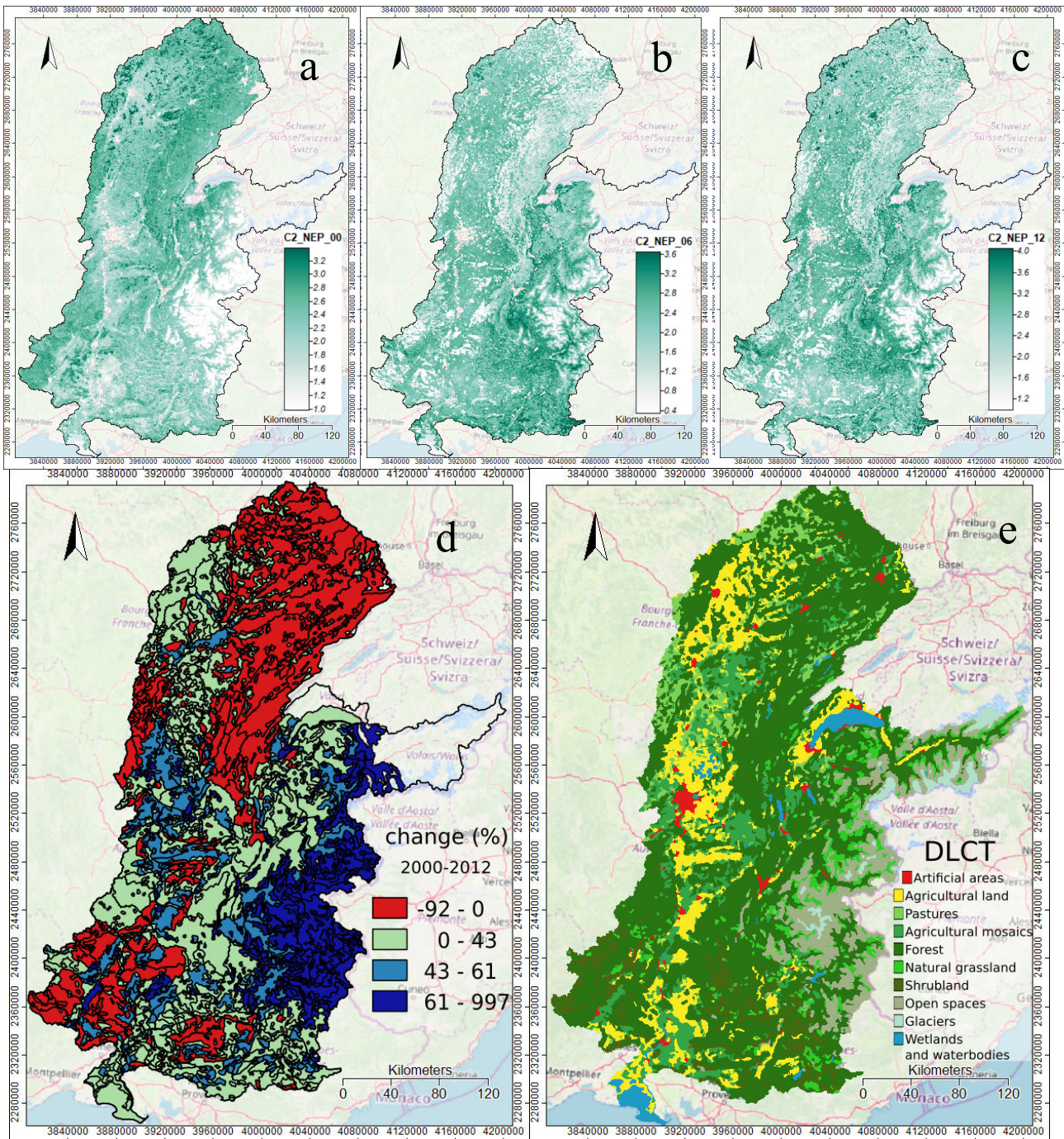


Figure 3.15. Carbon flow. Net Ecosystem Production (tC ha⁻¹) calculated as the difference between NPP and respiration in 2000 (a), 2006 (b), and 2012 (c). (d) shows the change (%) from 2000 to 2012, the information in transboundary sub-basins relies only on data in the French part. The cool colors indicate an increase on flow. (e) illustrates for comparison the Dominant Land Cover Types in 2000. QGIS and SAGA-GIS geo-processing tools were used.

Chapter 3 Ecosystem Carbon Account

Improvement of the present picture will require in the first place data on secondary carbon stocks, inflows from other countries, consumption and production returns, and dumping leakage and emissions to biosphere. Due to the lack of these data the total inflow of bio-carbon in the Rhône watershed equals the Net Ecosystem Production.

The informations analysed here represent five of eleven terrestrial reservoirs in the IPCC simplified scheme of the global carbon cycle. These five reservoirs or carbon stocks contain 85% of the total terrestrial carbon cycle due to the higher concentration of carbon in soils.

3.3.3 Use

The average use of bio-carbon through agriculture harvested crops is 2.58 tC ha⁻¹ with SD of 1.14 tC ha⁻¹ in 2000, 2.43 tC ha⁻¹ with SD of 1.15 tC ha⁻¹ in 2006, and 2.52 tC ha⁻¹ with SD of 1.15 tC ha⁻¹ in 2012. The smaller value in 2006 is due to a lower yield of cereals, forage and prairies. This could be related to a reduction of the Net Ecosystem Production in the same year.

The average wood removals is 0.36 tC ha⁻¹ with SD of 0.44 tC ha⁻¹ in 2000, 0.36 tC ha⁻¹ with SD of 0.32 tC ha⁻¹ in 2006, and 0.30 tC ha⁻¹ with SD of 0.24 tC ha⁻¹ in 2012.

For withdrawal of animal bio-carbon, in the absence of data on equidae and poultry, the average is 0.42 tC ha⁻¹ with SD of 0.19 tC ha⁻¹ in 2000, 0.33 tC ha⁻¹ with SD of 0.15 tC ha⁻¹ in 2006, and 0.33 tC ha⁻¹ with SD of 0.16 tC ha⁻¹ in 2012.

The total use of bio-carbon is obtained by the addition of agriculture, wood removals, and withdrawal of animal bio-carbon, to give on average 1.05 tC ha⁻¹ with SD of 0.87 tC ha⁻¹ in 2000, 1.02 tC ha⁻¹ with SD of 0.79 tC ha⁻¹ in 2006, and 0.96 tC ha⁻¹ with SD of 0.81 tC ha⁻¹ in 2012. The largest proportion of biomass extraction occurred in the low Saone watershed, an important crop area. Areas of timber extraction are noticeable in the previous Franche-Comté region and in the northern part of the Rhône watershed (**Figure 3.16a, b, c**). The rate of change is more prominent in the south east part of the basin for forest in particular (**Figure 3.16d, e**).

The total use in the Rhône watershed of crop, livestock, and forest biomass in the Rhône watershed in 2006 is around 6,000,000 tC or 179,502,902 of fresh weight for a population of 9,000,000 people. For comparison, in British Columbia, Canada's province, it was approximately 50,000,000 tonnes of fresh weight of crops, livestock and poultry, forestry, milk, maple and honey, and fisheries for a population of 5,000,000 people. The highest biomass extraction occurred through forestry (MEGS, 2013).

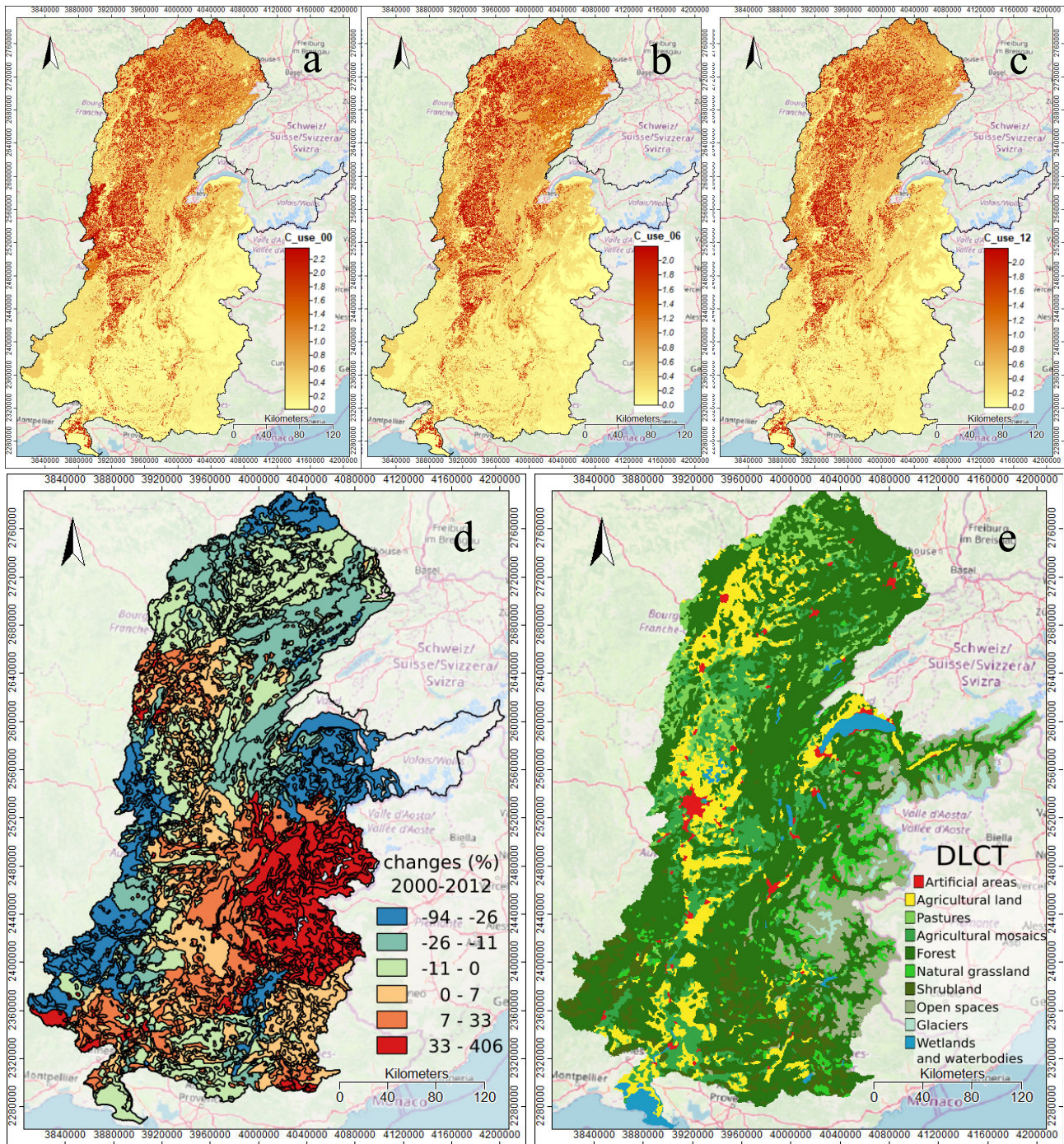


Figure 3.16. Carbon use ($tC\ ha^{-1}$) 2000 (a), 2006 (b), 2012 (c) calculate as the addition of agriculture harvested crops (Figure 3.10), wood removals (Figure 3.11), and cattle (Figure 3.12). 95% of data are between 0 (white) to 2 (darkest color). (d) shows the percentage of change from 2000 to 2012, the information in transboundary sub-basins relies only on data in the French part. The warm colors mean an increase in the use. (e) illustrates for comparison the Dominant Land Cover Types in 2000. QGIS and SAGA-GIS geo-processing tools were used.

Chapter 3 Ecosystem Carbon Account

Finally, the total annual losses of carbon due to land use change were 813,636.70 tC yr⁻¹, meaning in average 63.67 tC ha⁻¹ yr⁻¹. 97.28% of the losses correspond to the conversion from agricultural and natural land to artificial land-cover, and 2.72% from natural land, mostly grassland, to agriculture.

Intensity of use

The intensity of use is calculated as Net Ecosystem Production (corresponding to net ecosystem accessible carbon) divided by use. The average index is 3.43 in 2000, 3.26 in 2006, and 4.40 in 2012, meaning that the accessible ecosystem production of carbon is more than three times higher than its use. However, in the upper part of the watershed, the index is frequently below one meaning that the consumption of the ecosystem bio-carbon in the three years also affected the stocks (**Figure 3.17**).

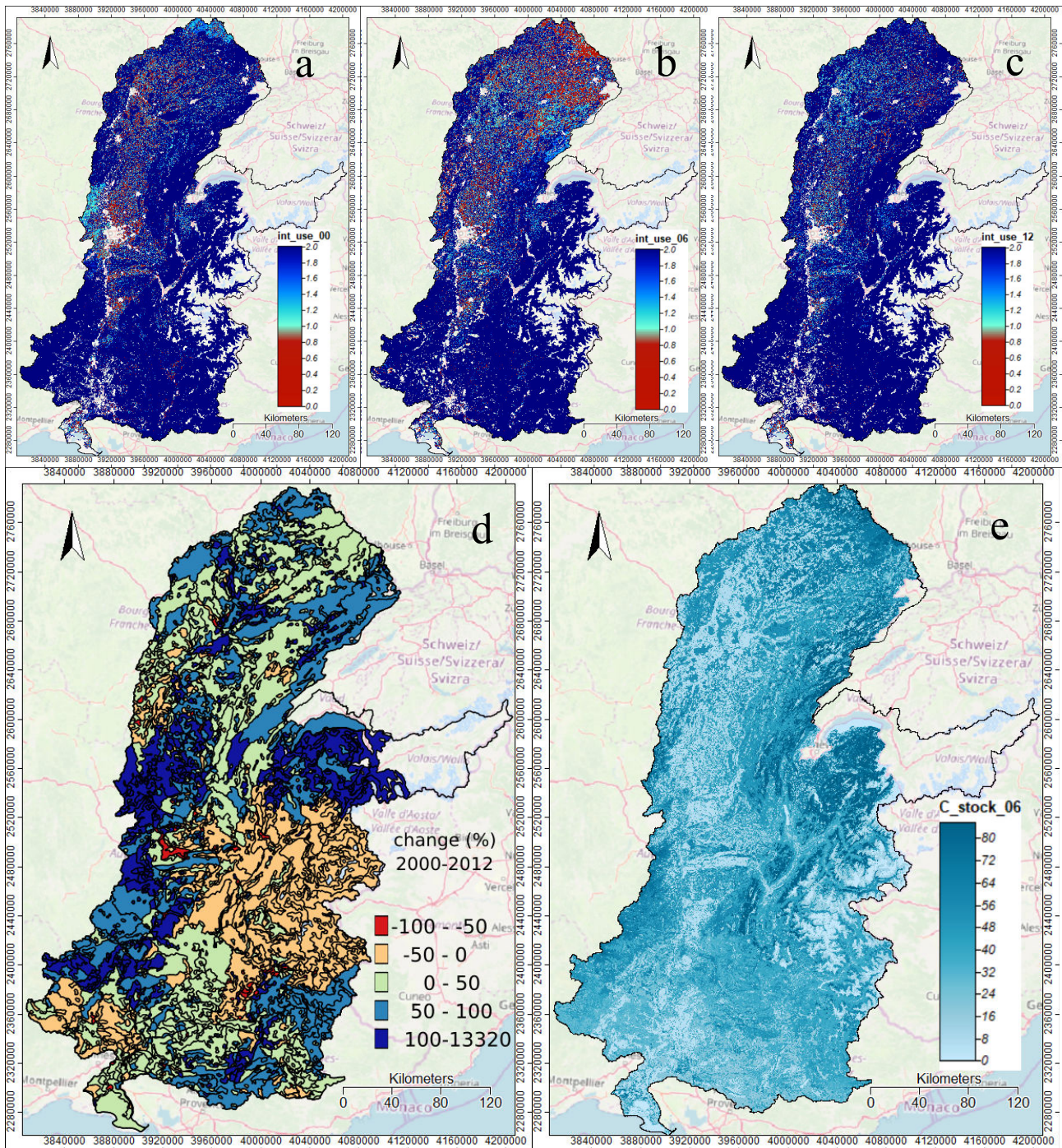
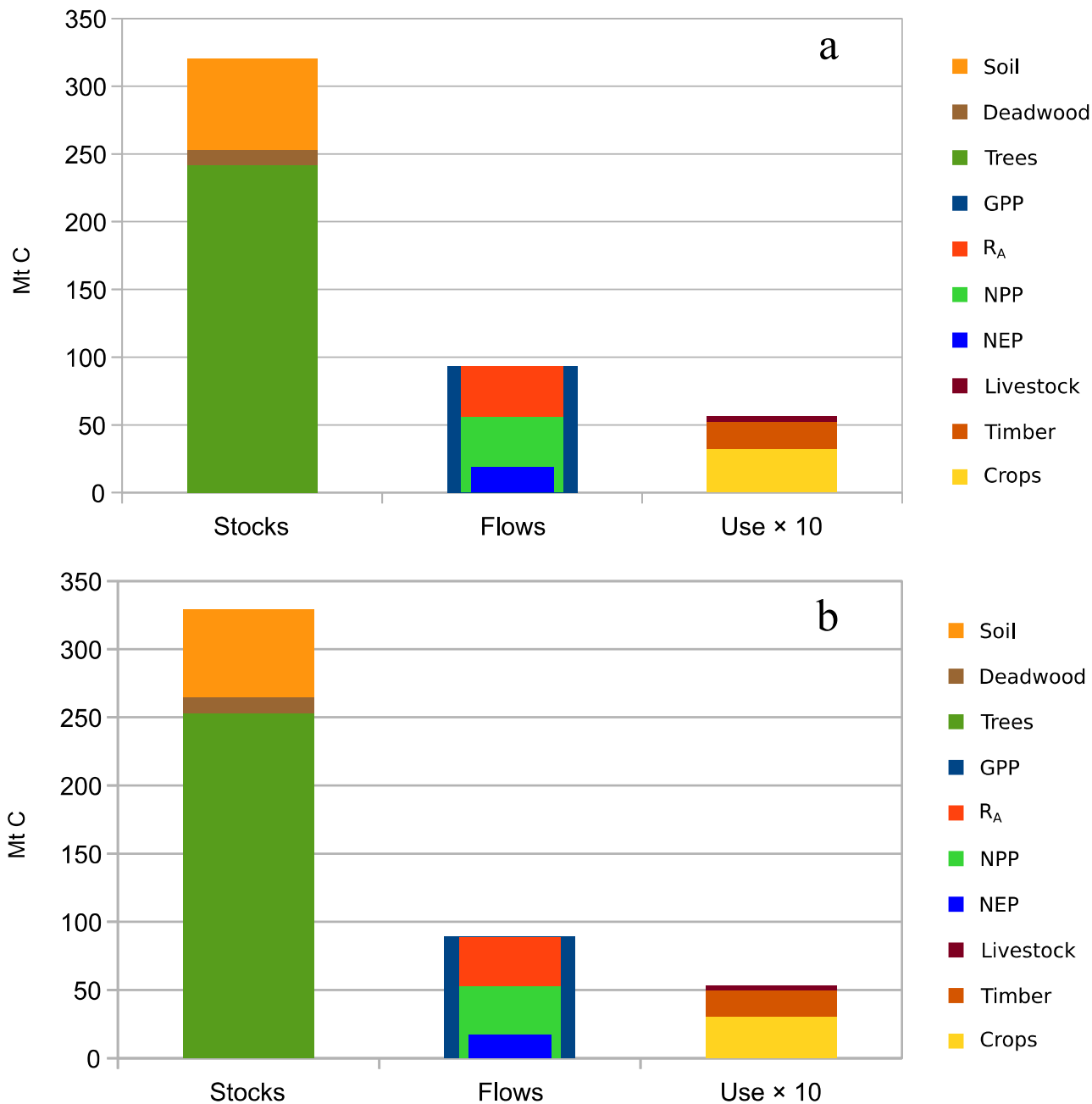


Figure 3.17. Carbon intensity of use 2000 (a), 2006 (b), and 2012 (c) calculated as NEP (Figure 3.15) divided by Use (Figure 3.16) values less than one indicating that the use exceeds ecosystem production. In 2006 there was a reduction in the production of the ecosystem which resulted in an increase in the intensity of the use. (d) shows the change (%) from 2000 to 2012, the information in transboundary subbasins relies only on data in the French part. Warm colors indicate an increase in the intensity of use. (e) represents for comparison the carbon stocks in 2006. QGIS and SAGA-GIS geo-processing tools were used.

Chapter 3 Ecosystem Carbon Account

In summary, trees are the main carbon stock, with only 20% of the GPP remaining as NEP in the flow, and the use corresponding to 33% of the NEP in 2000, 32% in 2006, and 42% in 2012 (Figure 3.18).



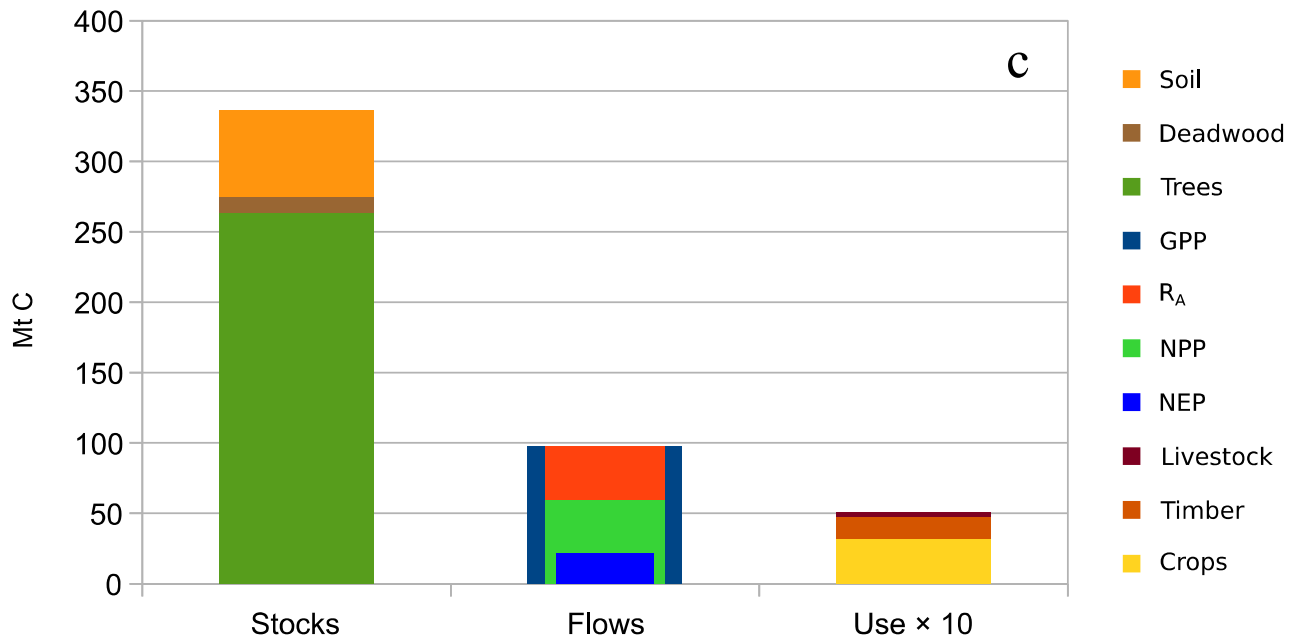


Figure 3.18. Carbon stocks, flow and use 2000 (a), 2006 (b), 2012 (c).

3.4 Main issues

Our work identifies data sources as the most critical obstacle in implementing ENCA with sufficient accuracy.

The scale of work (1 ha) limited the use of national data to three of 35 available sources.

Overall, I was able to analyse 10 of 14 main carbon accounting items. Even those ten have their own limitations. For example, data reporting campaigns were not consistent for most of the variables and this is particularly true as far as time series are concerned. In addition, to match the carbon accounts with CORINE (2017), I had to limit the analysis to three years.

Also the SD figures are frequently high and therefore there is a need to investigate how primary providers generate and process the data.

Error sources should be considered for each *geoprocessing* operation. For example, the interpolation process used to convert the point or polygons data sets into raster format can give highly erroneous outputs (Siska and Hung, 2001).

3.4.1 Stocks

Bio-carbon in aboveground living biomass

In the case of the official statistics overestimation could be done due to the mixed surfaces and the age of forest. The different campaign years by department have a confidence interval < 30% for forested area and < 80% for tree volume.

For the vector data base of forested area, the Gard department data was not available in the most recent version (from 2007 to present), so I was limited to the data from the previous version to complete the Rhône watershed account.

Bio-carbon in litter and deadwood

Only information at regional scale was available with dates poorly specified and with the same confidence interval as above. The wood density of living trees was considered to perform the calculations, knowing that the density of the deadwood would decrease as a function of time and species.

Bio-carbon in soil

Two sets of data from different sources were analysed. The results differ by one order of magnitude. The estimation based on European data sets (ESDAC) covered the entire Rhône watershed (9,112,168 ha), had an average of 75.90 tC ha⁻¹ with SD of 26.89, minima and maxima values of zero and 179.60, respectively, and a total amount of carbon of 648,882,675.59 tonnes. The GISSOL

dataset covered 8,549,277 ha, had an average of minima and maxima values of zero and 313.09, respectively, and a total of 68,248,517.55 tC for the French part of the watershed. The INRA team reported that 89% of the results were in the confidence interval and the errors at a depth of 30 cm were the lowest, namely 2 g/kg. I have chosen the INRA model because carbon estimates are soil sample-based and seem to have reduced levels of error.

An important issue would consist in attempting to explain the reason of the ten-fold difference between two models used.

3.4.2 Flows

Net ecosystem carbon balance is the carbon accumulation by an ecosystem. To generate the full picture for the watershed, additional data is needed on carbon loss, respiration and leaching, disturbances, increase of secondary stock, inflows from other countries, and production and consumption returns to the ecosystem. For all these variables the data when available is not at the required scale.

Gross Primary Production (GPP)

The computation of the annual sums required the use of specific software and programming skills due to the amount and format of data.

Net Primary Production (NPP)

The NASA time series raised problems for the period 2000-2003 due to the change of land cover data system in the model (Running and Zhao, 2017). This inconsistency had as consequence negative values for the autotrophic respiration in the interval 2000-2003. For that reason, the data for 2000 was estimated roughly.

The data information from the DLR is at a higher resolution as compared to NASA data. Since the GPP data from the DLR was not available, I was not able to estimate the autotrophic respiration and the Net Ecosystem Production based on DLR values.

Autotrophic (R_A) and heterotrophic respiration (R_H)

For the two respiration parameters I made raw calculations according to available data. More complete information is required for the respiratory component to more precisely evaluate the cost of maintaining and producing plant and animal biomass.

3.4.3 Use

Bio-carbon use data are provided by three sources (e.g., agricultural crops, livestock, and forestry) and generally lack spatial information and time series at the Rhône watershed scale. Biomass extraction from fisheries and poultry was simply not available resulting in an incomplete evaluation of bio-carbon use.

Agriculture harvested crops

The vectors files for 2000 and 2006 were not available for the Rhône watershed. I used the 2012 data instead for the three years: there were 1,451,934 polygons of crops making up 3,278,443 ha of agricultural surface. The official statistics for 2012 cover 1,249,852 ha as compared to 1,249,224 for 2006, and 1,254,126 for 2000. In consequence the spatialization of bio-carbon based on the official statistics was not possible for approximately 60% of the area in the three years.

Wood removals

The units for values of data series from AGRESTE DISAR changed from hundreds to hundred thousands from 2012 to 2013 with no warning to the users and no explanation on the method of calculation. Nevertheless the series were used from 2005 and 2012 and completed with different data set for 2000 to 2004.

Withdrawals of animal bio-carbon

The data about fisheries was not available, except at national level (AGRESTE, 2018).

All in all, I estimate that the resolution of the carbon account for the 25% of information analysed here could be improved by a factor of 1.2 to 1.5 in the event of managing the above obstacles and considerations.

Conclusion

In this chapter I provide the first exhaustive picture of the bio-carbon state in the analysed territory. Due to the lack of sufficient and more elaborated information, the Net Ecosystem Production was employed to determine the Accessible resource. With the available information, the estimation of the index of intensity of use of bio-carbon showed that agricultural production in some areas in the Saone river basin is not sustainable. This index will serve to the further calculation of Carbon Unit Value and its integration to the Ecosystem Capability Unit in Chapter 6.

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Chapter 3 Ecosystem Carbon Account

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Chapter 4 Ecosystem water account

Outline

The chapter deals with the Water and River accounts, a major of the three core accounts in ENCA. A broad range of variables on stock, flow, and use were considered and analysed, and illustrated with an overview of accounting tables that have been designed for generating the results. The chapter focuses on the specification of definition and classification of the statistical and hydrological units, the presentation of the accounting framework and is reporting only selected parts of the full account. The results concentrate on a couple of key indexes that characterize the ecological status of the watershed.

Acknowledgement

The hereafter water and river accounts have been produced through a collective enterprise. These accounts represent the most complex, conceptually and technically, component of the ENCA approach.

The conceptual dimension and the working guidelines as outlined in the ENCA-QSP handbook (Weber, 2014) have been adapted to the ENCA Rhône protocol by Jean-Louis Weber. He has produced a detailed report bearing the title *Rhône river catchment: accounting for the water resource and the ecological condition of rivers*.

The heavy methodological section has been synthesized and edited by myself, with a constant back-and-forth input (data research, data processing, and data sheets elaboration) from the EVS Laboratory, Hervé Parmentier and Luc Merchez in particular. The complete methodology is accessible via the link².

Only information and results – to which I contributed, that pertain to the production of the ENCA proof-of-concept have been incorporated in the thesis manuscript. They are not commented in detail here because they only make entire sense in the frame of the complete water and river accounts.

² <https://docs.google.com/document/d/1ft-R-kpzftTs2Nwu9YaEBEGYwyig2WTuQZWe4frVbY/edit?usp=sharing>

4.1 Introduction - Water environmental accounts

In 2003 the United Nations' System of Environmental-Economic Accounting (SEEA) introduced water accounts which were upgraded in 2009 in an interim manual known as the SEEA Water. Main chapters of this manual are summarised now in the SEEA Central Framework 2012.

Although inspired by the French Natural Patrimony Accounts (NPA) tested in the 80's to which they borrow several features, the SEEA water accounts adopt a significantly different approach. While the NPA developed a systemic approach, the SEEA puts the focus on water supply and use for economic sectors (Vardon et al, 2007), neglecting *de facto* important quantitative and qualitative aspects of the water resource itself, such as its location, interaction between surface and groundwater, and water quality. The system approach to water accounts of early applications has been abandoned in most exercises where “flows” relate to uses of water by economic sectors and “assets”³ are generally limited to a quick assessment of volumes of surface water and groundwater. Out of the dozens of SEEA water experiments done by countries, a few have more visibility. They are all produced by statistical offices, some of them in conjunction with water agencies. All have put an emphasis on Supply-and-Use of water by economic sectors, which is consistent with the reference to the SEEA-Water. They all, at various degrees, express concerns regarding “assets” accounts, in particular regarding the definition of “stocks” of groundwater and rivers. As a consequence, the linkage between resource and use is rarely clearly done.

Australia⁴: biennial publication with very detailed breakdown of uses by sectors in support to the national policy of allocation water abstraction rights by tendering.

South Africa⁵: a first generation of accounts not continued up to a recent new attempt of water quality accounts in the context of the NCA&VES project.

Canada⁶: several partial accounts with focus on water supply and use. Issue for compiling assets accounts have been caused by the unclear definition of “stocks”. Statistics Canada proposes to redefine stock as renewable resource, a solution taken by Brazil (and followed for ENCA-Rhône).

3 In principle, SEEA Water accounts are lakes and reservoirs, rivers, groundwater, snow and glaciers and soil water.

4 Australian Bureau of Statistics, 2019, *Water Account, Australia, 2016-17*, <https://www.abs.gov.au/AUSSTATS/abs@.nsf/mf/4610.0>

5 Statistics South Africa, 2006: *Natural resource accounts: Updated Water Accounts for South Africa: 2000*, Discussion Document - D0405, Pretoria, December, 2006

6 Statistics Canada, 2019: *Water Accounts and Earth observation in Canada: The potential and challenges*, System of Environmental Economic Accounting Fourteenth Meeting of the UNCEEA 24-25 June 2019 https://seea.un.org/sites/seea.un.org/files/seea_water_accounts_and_earth_observation_data_canada.pdf

Mexico⁷: in its long tradition of environmental accounting, Mexico publishes series of water accounts on a regular basis. They focus on supply and use and include hybrid tables presenting side by side physical and monetary data. Attempts have been recently done to connect water use (in particular irrigation) and stress on groundwater.

Brazil⁸: comprehensive water accounts with details on supply and use by sectors as well as measurement of water assets (considered as renewable resource). The accounts are produced by river basins and hydrological units.

Mauritius⁹: full set of accounts 2000-2013. They have been produced at the same time as the experimental ecosystem capital accounts and some data collected for the former have been used for the latter.

The Netherlands¹⁰: annual production of water accounts by small sub-basins. Focus on water supply and use (and waste water) with indication of the origin (surface water and groundwater).

At Eurostat, water statistics are compiled at the national level, with details on uses by economic sectors, with resources limited to rainfall and evapotranspiration adjusted from transfers between countries.

Important work on water accounting has been carried out at the European Environment Agency, resulting in particular in the creation of the European database of catchments and rivers called ECRINS, in the modelling of rivers runoff, and in the assessment for the European Commission of the use of water by economic sectors with some geographical breakdowns and a database with the results¹¹. In parallel and integrated to the project a map of soil humidity deficit was produced by modelling at the scale of 1 km² grid cells of meteorological data assimilated in the swbEWA¹² database.

7 National Institute of Statistics and Geography (INEGI) of Mexico, 2012, *Technical Note on the Water Accounts of Mexico*, UNSD, New York, https://seea.un.org/sites/seea.un.org/files/8_9.pdf et Francisco Guillén Martín (INEGI) 2019, *Policy applications of water accounts in Mexico*, UNECE Geneva, 2019, https://www.unece.org/fileadmin/DAM/stats/documents/ece/ces/ge.33/2019/mtg1/S2_3_Water_Accounts_Mexico_EN.pdf

8 Brazilian Institute of Geography and Statistics (IBGE), 2018, *Contas econômicas ambientais da água: Brasil 2013-2015*, https://biblioteca.ibge.gov.br/visualizacao/livros/liv101555_informativo.pdf and SEEA, 2018, <https://seea.un.org/news/brazil-responds-water-shortages-seea-water-accounts>

9 Statistics Mauritius, Ministry of Finance and Economic Development, 2015, *Water Account of Mauritius 2013*, <http://statsmauritius.govmu.org/English/StatsbySubj/Documents/quality%20of%20life/Water%20Account,%20Mauritius%202013.pdf>

10 Central Bureau of Statistics of the Netherlands (CBS), 2017, *Physical water accounts for the Netherlands* <https://www.cbs.nl/en-gb/background/2017/38/physical-water-accounts-for-the-netherlands>

11 A complete presentation of the work done can be found in a series of 4 reports produced in 2012 by POYRI, VITO and the EEA and sponsored by the European Commission. They can be downloaded from this site: <http://ec.europa.eu/environment/water/blueprint/balances.htm>. Findings are summarized in Crouzet P., 2013, *Results and lessons from implementing the Water Assets Accounts in the EEA area*. EEA Technical report No 7/2013

12 The swbEWA water balance model (Kurnik et al., 2014) is based on the E-OBS climate datasets produced for the EU-FP6 ENSEMBLES project (<http://ensembles-eu.metoffice.com>) those of the data providers of the ECA&D project (<http://www.ecad.eu>) (van den Besselaar et al., 2011).

Kurnik, B., et al., 2014, 'Analyzing seasonal differences between soil and water balance model and in-situ soil moisture measurements at nine locations across Europe', *Journal of Environmental modelling & Assessment* 19 (1), pp. 19-34.

Chapter 4 Ecosystem Water Account

The Rhône ENCA project started with the assumption that these databases could be reused, as were reused Corine land cover and derived applications. It is what happened with swbEWA which provided monthly data 2000-2016 on precipitations, actual evapotranspiration, and deep percolation which were used for accounting. However, the other data could not be used as expected because of scale issues regarding water uses supplied by river basins or of the quality of ECRINS where too many gaps with national data (rivers) and geometric errors (both for rivers and catchments).

As water is a key component of ecosystems, SEEA-Water concepts have been revisited in ENCA, contrarily to the SEEA Experimental Ecosystem Accounting framework where the status of water is unclear due to the assumption that the SEEA-Water covers the subject.

4.2 Main features of ENCA Water and Rivers accounts

The general circulation of water throughout nature and the human systems results in the multiplicity of actors and of a somehow fragmented knowledge. This is reflected in France by the multiplicity of institutions and economic actors involved at one or the other stage of water management, exploitation and use, and holding one part of the data. Efforts have been made in the last decades to come to a more holistic knowledge needed for more consistent water policies. Important milestones are the creation of the water agencies, the structuring and development of the national information system with achievements such as *inter alia* the BD-Carthage, the SANDRE, the ONEMA, and the national portal to provide widespread access to data on water (namely EauFrance). In the last period, the merging of ONEMA into the *Agence Française de la Biodiversité* (AFB) and the process initiated by the EU Water Framework Directive (WFD) evidence the important shift to a more integrated ecological approach to water issues.

Water accounts can be seen as a building block in the emerging information system with the purposes of synthesizing the measurements of the water resource and its use, as well as understanding water in the broader socio-ecological sense, which is the purpose of ENCA-QSP. Water and rivers accounts' purpose is not to produce new data on water, but to take advantage of the wealth of data and knowledge produced by the scientific, engineer, administrative, and business communities in charge of the many aspects of water. Accounts using their data may highlight some gaps due to different initial purposes. They may fill such gaps on a provisional basis, but never be a

substitute data validated by the water communities on the basis of their theoretical as well as empirical knowledge. ENCA-QSP addresses water as one of its basic modules, water being an essential part of the assessment of ecosystems potential and resilience. In terms of potential, the water resource available to humans as well as to nature is a central measurement issue. In terms of ecosystem resilience, intensity of human consumption of water, impacts of water pollution, and infrastructure restructuring are key variables to consider. Water accounts therefore play an important role in ecosystem natural capital accounting.

4.2.1 The system analysis behind the accounts

In his “Essay on the œconomy”, Pierre Calame tells that land cover is the skin of Earth. Following the metaphor, we can depict the hydrologic system as Earth’s blood circulation system. While land systems and their elements are to a large extent fix, the water system is characterised by the flows which connect its components and the latter to the land units that they water. The definition of the statistical units for water and rivers accounting is therefore critical.

In ENCA, water accounts (**Figure 4.1**) are established for assessing the condition of socio-ecological landscape units (SELU). SELUs are defined taking into consideration the characteristics of surface water systems (i.e. water circulation within river sub-catchments) and dominant land cover characteristics. The *quantitative water accounts* record exchanges between the hydrological system units and of the latter with the use system of water withdrawals, consumption, and returns. *Water quality accounts* are compiled, in particular on the basis of the information reported to the EU Water Framework Directive, regarding water quality status and the good ecological status of rivers.

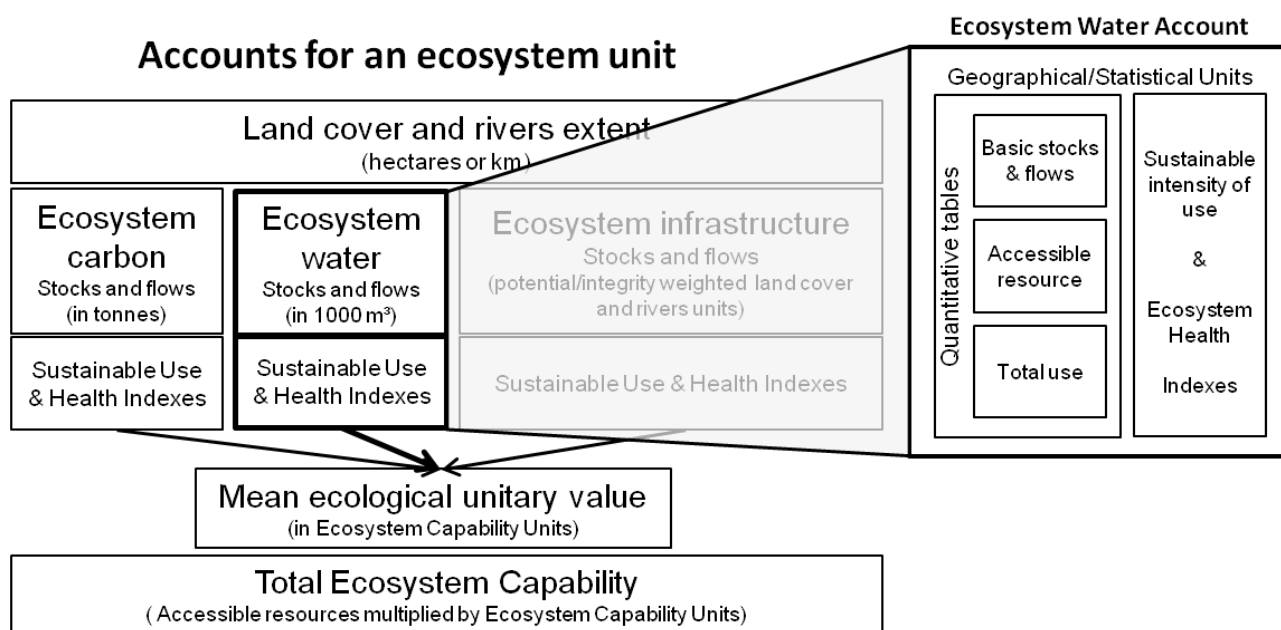


Figure 4.1. Ecosystem Water Account in the ENCA framework (rectangles in bold).

The ENCA *Water accounts* are based on the same system analysis as that defined by Jean Margat in *Les comptes du patrimoine naturel* (NPS, 1986) and developments in Spain, Chile or the Republic of Moldova. The scheme below (**Figure 4.2**) is extracted from the Moldavian report¹³ on water accounts. In this model, the system analysis distinguishes for a territory of reference (typically a river catchment) a Users System which is connected to the Hydrological System in terms of abstraction and returns of “blue water”. Therefore, *in situ* uses and consumption (in particular evapotranspiration of rainfed and irrigated agriculture) take place in the Hydrological System. This scheme has been modified in the SEEA-Water where it makes a distinction between “natural system” and “economic system”. This modification is motivated by the presentation of “supply-and-use” tables which consider products in the economic sense, and not in their substance. One consequence is that the SEEA-Water and Central Framework have introduced the calculation of water used by rainfed agriculture as an abstraction of rainwater (followed by evapotranspiration and returns), while in ENCA-QSP, this is considered spontaneous evapotranspiration.

13 Inland Water Accounts of the Republic of Moldova. Preliminary Results of Resource Accounts in Raw Quantities, 1994 & 1998. First Elements for Environmental Expenditure Accounts for Protection and Efficient Use of Water, (2000), Report for Eurostat prepared by Jana Tafi (DASS/Moldova) & Jean-Louis Weber (IFEN/France)
http://www.ecosystemaccounting.net/?page_id=2

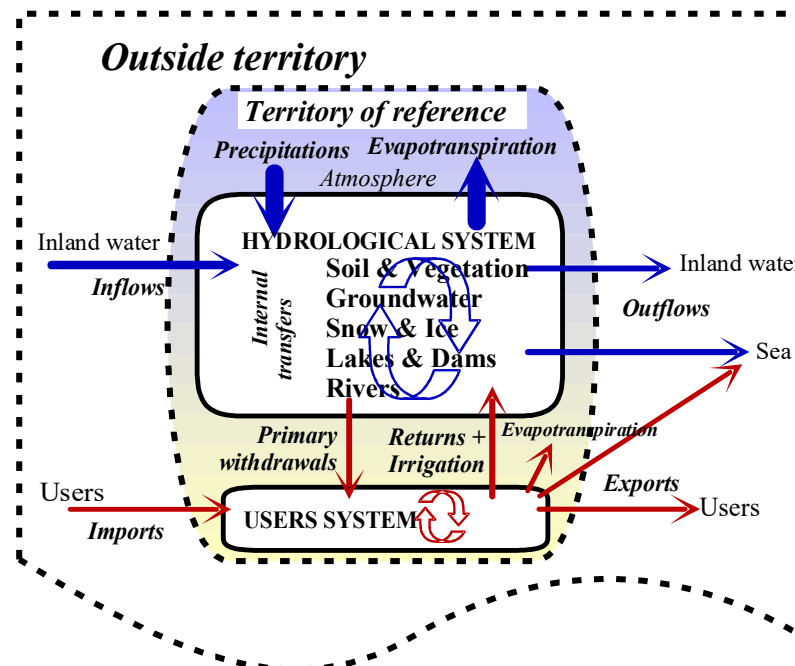


Figure 4.2. The water accounts model and main water categories. *The different categories of water used in the ENCA are defined below. **Blue water**: water that can be withdrawn for use; it includes natural inputs and post-use returns. **Green water**: rainwater that is used by vegetation (especially agriculture and forestry); it is measured by the corresponding spontaneous Real Evapotranspiration (total REE minus irrigation-induced REE). **Brown water**: wastewater; it can be subdivided according to pollutant concentrations; this is not currently done in ENCA. Brown water is measured in m³ of water and tonnes of biocarbon (carbon account: carbon returns to rivers, to the ground (spreading) or to the atmosphere (incineration)). **Grey water**: water from rivers and lakes that cannot be used due to environmental constraints: instream flows (*débits réservés*), maximum biochemical oxygen demand concentrations.*

NB: SEEA-CF and Water add green water to blue water, which is contrary to the usual rules of hydrology. The reason is that SEEA water is a product, while in ENCA it is the component of the system. Grey water is also ignored by the SEEA, as it is used *in situ*. In practice, SEEA Water applications do not respect the prescribed methodology and do not address these details.

In addition to water accounts ENCA produces **river accounts**. They are closely connected to water accounts through the measurement of river reaches in standard river kilometres (SRKM) which are defined as the product of their length by the discharge. $1 \text{ srkm} = 1 \text{ km} \times (1 \text{ m}^3/\text{sec})$ (see **Methods section 4.4**). SRKM weighted rivers are comparable whatever their size and can be aggregated, operation which is disputable when quality is referred to lengths only (as it is for maps). The integration of landscapes and river systems in SELUs makes it possible to combine their assessment in the account of infrastructure integrity and biodiversity (see **Chapter 5**).

The water accounts undertaken in the context of the ENCA Rhône project, this report covers the French part of the watershed (**Figure 4.3**). Although data are abundant for the Swiss part, important

elements used for France, namely the detailed reporting to the EU Water Framework Directive were not available.

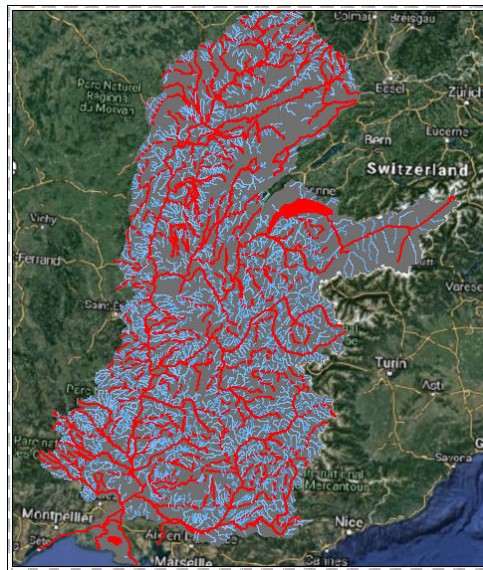


Figure 4.3. Rhône river watershed and catchment's main rivers

4.2.2 Definition and classification of statistical units for accounting

The integration of main parameters, data types and their relationship as explored during this work are presented in the diagram below (**Figure 4.4**).

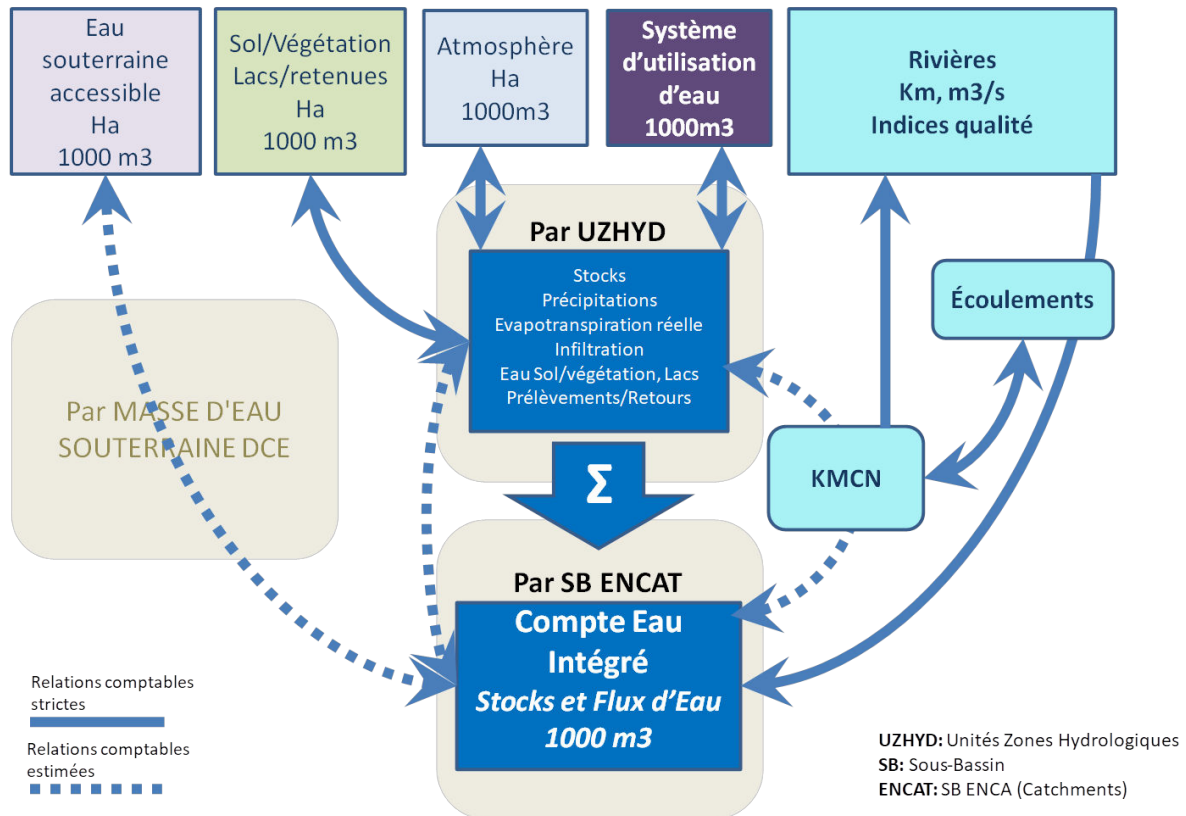


Figure 4.4. Statistic units at different scales and integration of data for the Rhône Water and River account (in French).

Surface hydrological accounting units have been defined for the whole Rhône river catchment using French data from BD Carthage and SANDRE, and Swiss data from OFEV.

River classification

Rivers are classified in 3 levels (**Figure 4.5**) derived from their Strahler level. Class 1 is the Rhône river itself, class 2 its main tributaries and so on. Classes 6 (small streams) and 7 (very small streams) have been merged as the latter are poorly mapped in BD Cartage while absent from the OFEV layers. Because OFEV covers the whole Lemman sub-basin, it was possible to establish a correspondence between Swiss and French classifications.

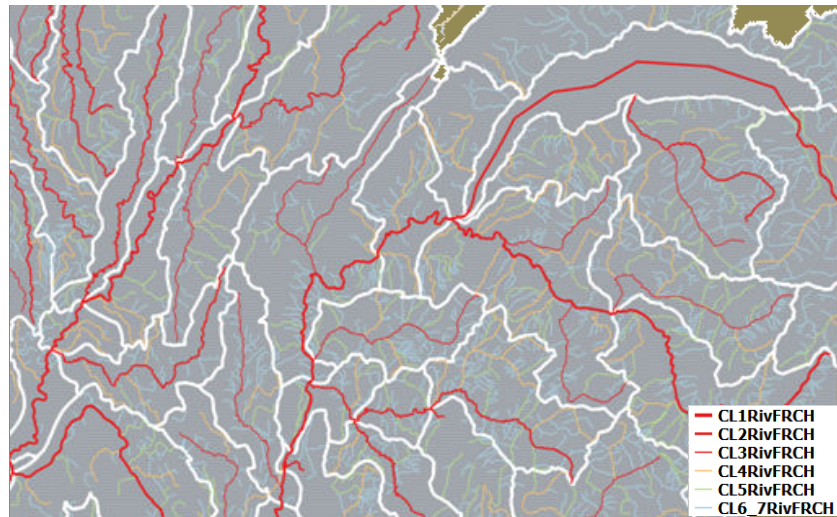
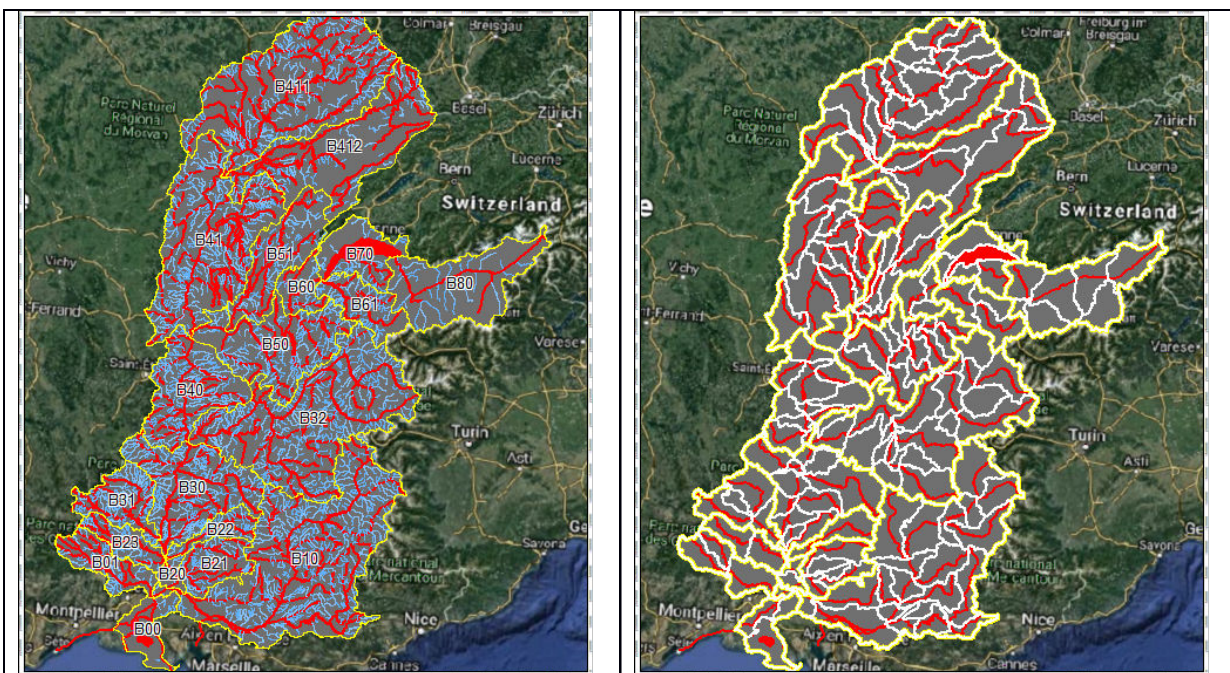


Figure 4.5. Extract from the map of rivers with color codes for each class.

Definition and classification of catchment units

Complete water accounts are established for ENCATs (ENca CATCHment units) (Figure 4.6), the sub-basin units being used for *integrating land and rivers accounts*. They can be aggregated for higher level catchment units such as the Sub-Basins zonings in use at the national and European scales (coded SB). ENCATs are the aggregation of hydrological units (UZHYD), but full accounts are not established at this stage at this scale. However, key outcomes of accounting variables are downscaled by UZHYD, as well as by Socio-ecological landscape units (SELU). There are in total 164 ENCAT units and 649 UZHYD.



<i>Coding of SBs</i>	<i>ENCAT are embedded into SB's boundaries</i>
<p>Sub-basins (SB or B) are given a code according to the connection of their main river drain in the tree network of the basin. SB code second digit with 0 means that the main drain in terms of runoff is the Rhône river itself (from B00 to B80). Tributaries basins are coded according firstly to their exit, a second digit being an ID (1, 2 or 3).</p> <p>ENCAT are exact subdivisions of SBs. They are given a tree code which is composed firstly of the SB code followed by a number defined in an incremental way. For example, B10_112 designates an ENCAT flowing into an ENCAT coded B10_11. The latter is a tributary of B10_1 which is the first ENCAT of B10.</p> <p>This incremental way of coding has been borrowed to that of FECs in ECRINS. However, there is a difference as in ECRINS, FECs coding does not reflect belonging to Sub-Basins as the latter are not tree-coded.</p>	

4.2.3 Hydrological units and other zonings of land

Land cover features

In principle, land covered by freshwater should be taken into account in the making of water and river accounts. In practice, only lakes and glaciers have been processed separately. Rivers are recorded as linear features to which are attached data on runoff. Wetlands (in the senses of CORINE land cover, i.e. marshes), flooded agriculture (rice), and transitional waters have not benefited of a specific treatment. In the future, such information should be taken into account, for example on the basis of recent mapping of temporary water surfaces produced by the Joint Research Center (JRC) and by the COPERNICUS Global land service.

In terms of surface objects of particular importance for the Rhône ENCA test, lakes surface has been used to attempt an estimation of lakes and reservoirs volumes, an information poorly documented, although existing with the institutions and companies managing them.

Similarly, estimations of glaciers volumes is very fragmented, limited to a number of sampling points. After comparison of glacier maps and quality checks against satellite imagery, CORINE land cover proved to be the most comprehensive and accurate, while providing insights on the (dramatic) loss of surface, which was used for estimating volumes.

ENCAT and Dominant Land Cover Types

The ENCAT zoning has been calculated following the default methodology used for defining SELUs by the intersection of dominant land cover types (DLCT) and of boundaries of river basins (**Figure 4.8**). From this point of view, SBs are too large to discriminate SELUs, while the intersection of DLCT with UZHYD generates too many very small units of little ecological meaning.

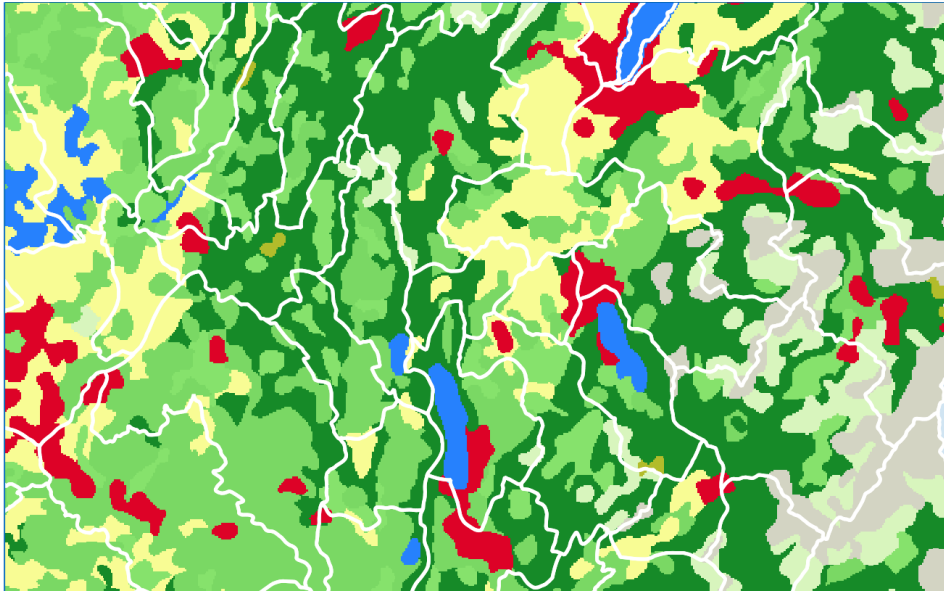


Figure 4.8. Intersection of DLCT and ENCAT (white lines) for producing SELU

Administrative and hydrological zoning

Another zoning that has to be considered is the administrative zoning, in particular the municipalities for which are available population census and other social and economic statistics. There are obvious issues of mismatch between municipalities and UZHYD boundaries. This point is well taken into consideration by databases on water withdrawals and waste water treatment plants which geo-reference data to UZHYD. This is not the case for other statistics and concerns all cases where a municipality belongs to two or more UZHYD. On **Figure 4.9**, municipalities are mapped with yellow lines and UZHYD boundaries in dark violet. These are mountain areas where relief (hence UZHYD boundaries) has shaped administrative divisions. Even in that case, intersected municipalities are numerous.

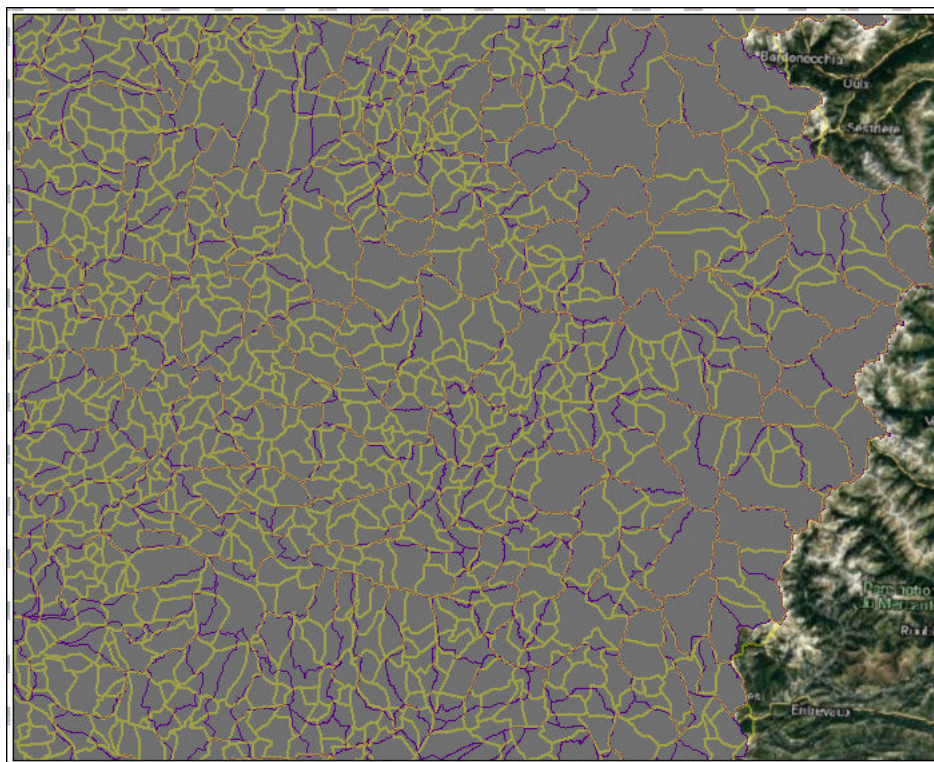


Figure 4.9. Municipalities (in yellow) and UZHYD (in violet).

4.2.4 Units for soil water and groundwater accounting

Soil mapping units

The soil units which are used for accounting for soil water are those defined in soil databases, in Europe by JRC's European Soil Data Centre (ESDAC) and in France by INRA and its GISSOL partnership. INRA is a key player in ESDAC. These data bases refer to Soil Typological Unit (STU) and the Soil Mapping Unit (SMU). Each SMU contains 1 to 10 STUs, for which the percentage of the SMU is known but not the position within the SMU. To SMU and STU are attached various attributes, either structural (soil density, stone content, depth with different layers; "horizons" of 30 cm, 1 m...) or estimations of carbon content, water content, biodiversity or erosion based on sampling and modelling. Therefore, ENCA needs to match as much as possible the patterns of soil data bases in order to enable the integration of its own accounting modules.

Groundwater bodies units

In ENCA, only the groundwater renewable resource is recorded with the purposes of (1) assessing the sustainable accessible resource by ENCAT and ultimately by SELUs and (2) assessing groundwater replenishment and quality status by SELU.

The SEEA is unclear on what a stock of groundwater is and how it has to be recorded in water accounts. The “stocks-flows-stocks” sequence is not very helpful as water tables pile-up and all cannot be considered as a resource. In ENCA-QSP, groundwater is considered only in its relation to surface water. This leads to proposing a concept of *sustainable accessible ground water* based on the assessment of groundwater net recharge. A given water table can be expressed as: deep percolation – draining to rivers – abstraction +/- exchanges with other tables. The solution adopted in ENCA is similar to that of IBGE water accounts in Brazil, as well as to what is envisaged by Statistics Canada. It is consistent with what is measured by hydro-geologists, i.e. the piezometric levels at given points (measured by piezometers or through the observation of water level in wells). From an ecosystem water accounting perspective, the question of the accessibility and sustainability of groundwater water for a given SELU is important. It requires at least three things, which are (1) mapping the interface between surface and groundwater tables, (2) taking into account groundwater circulation and exchanges, and (3) integrating observations on groundwater status (hopefully existing and accessible on line due to the EU WFD reporting).

Comprehensive sets of geo-data of groundwater tables maps exist in BDLISA (**Figure 4.10**), produced by the *Bureau de Recherches Géologiques et Minières* (BRGM). It includes a shapefile with the groundwater bodies used for reporting the EU WFD: “couches affleurantes”. Attributes of this layer are Quantity and Quality status for the years 2009 and 2015. However, this layer cannot be used in a simple way. First of all, groundwater bodies generally do not match river catchment boundaries, as shown in **Figure 4.10**.

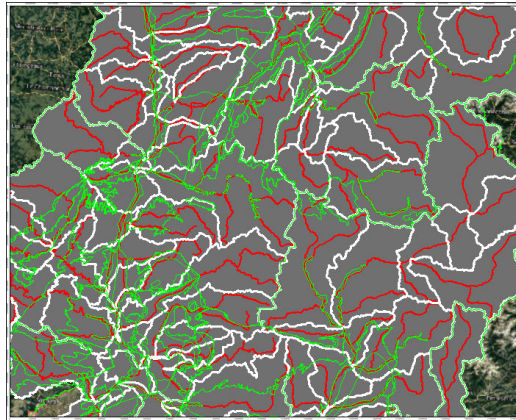


Figure 4.10. Map of BD LISA’s “couche affleurante” (green lines), ENCAT rivers (red lines), and ENCATs (white lines).

Secondly, water tables are generally larger by several orders of magnitude as compared to surface units such as SELUs. Thirdly, groundwater is not only made of surfaces, but mainly of volumes that exist at different depths, which means that water tables often overlay. A given water table can be deeper in some places and shallow in others, as shown in **Figure 4.11**.

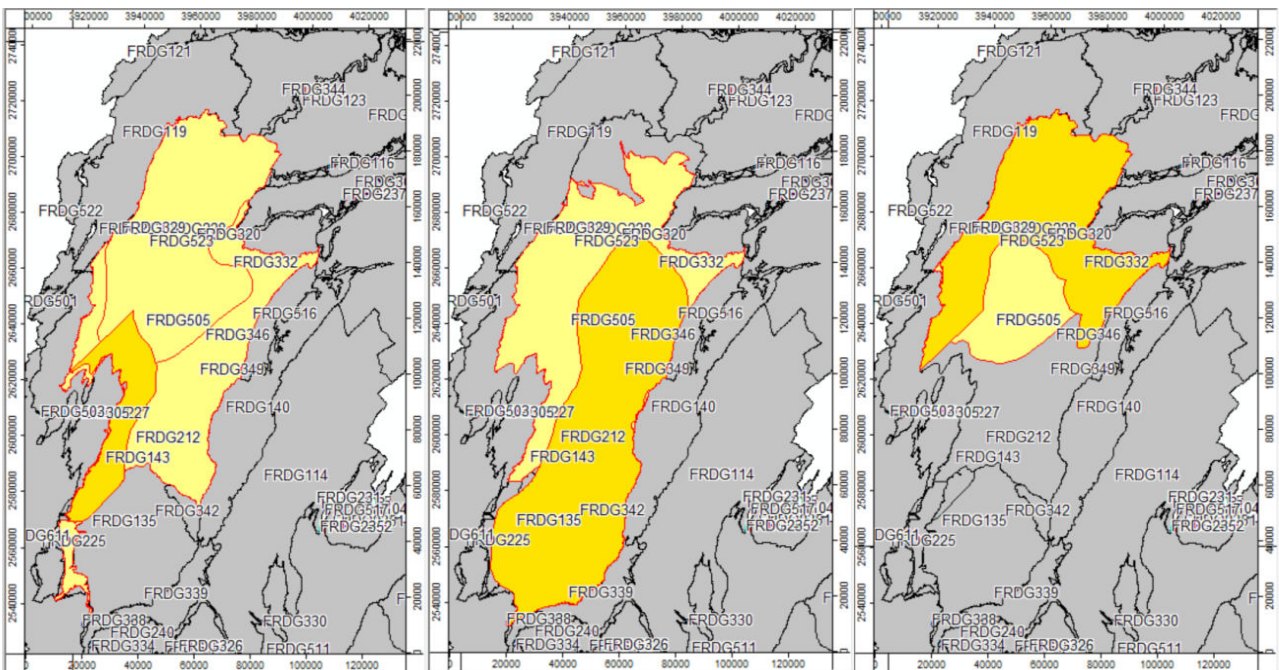


Figure 4.11. Overlaps of shallow water tables (“nappes affleurantes”). In this region, water tables overlaps are shown in dark yellow.

Fourthly, groundwater bodies are entities within which water circulates and which exchange between themselves. Contrarily to soil which is spatially highly variable and mapping units are small, resampling of a whole groundwater body to land units is not possible.

From the surface, only deep percolation, draining to rivers, and abstraction (and returns after use when it happens, a case not considered for the Rhône catchment test) can be mapped. Water accounts therefore will be based on the interface mapped in BDLISA. This means that the index of groundwater use intensity will not be computed directly from the accounts but will be based on external data, namely the quantitative status reported to the WFD. The extraction of data for the surface - ground water interface is done using SAGA-GIS gridding modules resulting in a layer of grid cell of 1 ha to which values are attributed.

4.3 The making of ENCA Water and Rivers accounts for the Rhône river catchment

4.3.1 The water accounting framework

The purpose of ENCA Water and River accounts for the Rhône River catchment is to carry out a proof-of-concept of the ENCA model and its feasibility at a scale where it has not been tested yet. As a matter of fact accounting is about summarizing and putting together existing data, not to create new data in domains covered by agencies and experts. As a rule, using primary monitoring data of elementary variables has been avoided as much as possible. Preference has been given to expert data of the domain. From this point of view the approach has similarities with the basic principles of reporting to the EU Water Framework Directive (WFD) on quantitative, chemical or ecological status of water bodies, or the Article 17 reporting to the EU NATURA2000 Directive.

The test covers accounts described in the Water accounts of the CBD TS77 Technical Series Chapter 6 (**Figure 4.12**).

Chapter 4 Ecosystem Water Account

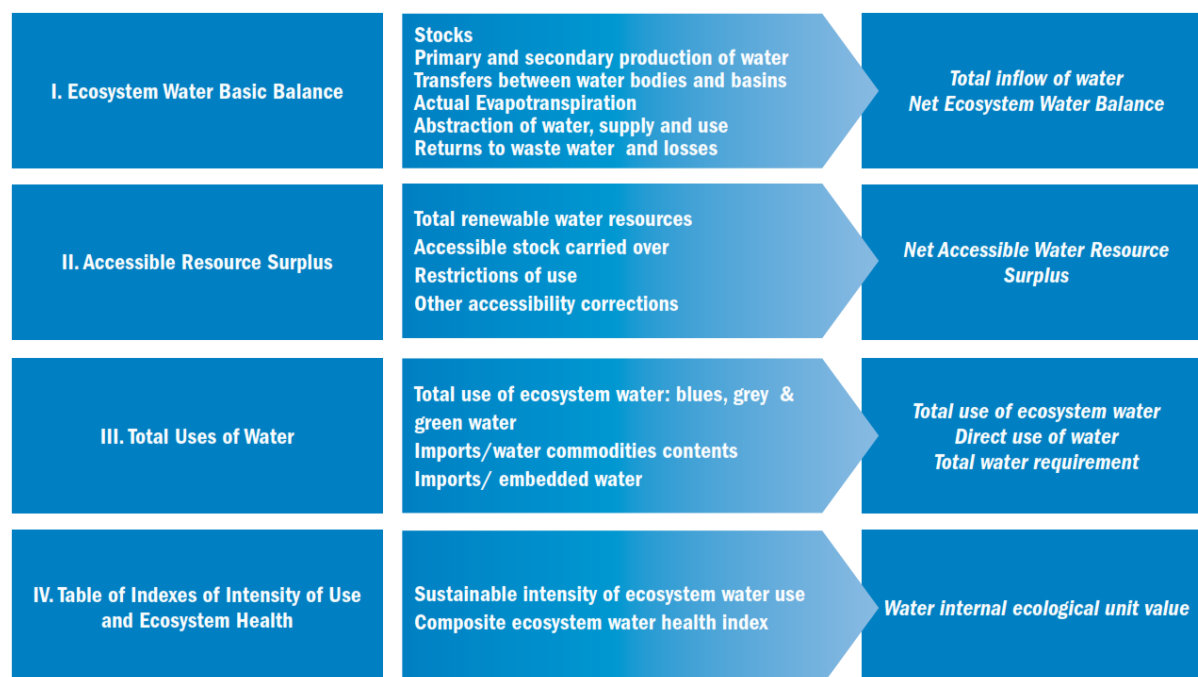


Figure 4.12. Structure of ENCA-QSP Water Account (Weber, 2014).

The water resource was compiled as assets by ENCAT and (in part) by UZHYD, and the water accounts reported by SELUs. Asset is the term used in the SEEA Water to name Lakes and Reservoirs, Rivers, Groundwater, Snow and Glaciers, and Soil Water. With the above mentioned difference regarding groundwater stocks, ENCA-QSP water accounts are largely consistent with the SEEA water. However, the classification of water uses is much simpler in ENCA, corresponding to data and categories available at the local scale.

The terms of reference of the project imply using the best data accessible in the basin and check their relevance, assess if they fit ENCA requirements and, if not, provide ways of improvement or alternatives. The candidate input datasets are very heterogeneous considering formats, acquisition methodology, spatial and temporal resolution, and intrinsic qualities. The ENCA data model is assimilating them into a common grid (here of 1 ha cells) before recombining data to feed the various accounting units defined in the **Section 4.2.2**.

Other data sources have been identified but not fully used due to the amount of work requested. It is for example the case of data from ESDAC or data from agriculture censuses and the “Registre Parcellaire Graphique” for agriculture. Simplified estimation methodologies have been used instead for the test.

In some cases, the best available data was simply not accessible at the time where a specific subaccount was processed. It is for example the case of the IRSTEA “maps of consensus” for rivers average and baseline runoff¹⁴ which have been used only for the estimation of groundwater draining to rivers. They were not available when computing the SRKM value of rivers, which required defining and using a second best methodology.

However, estimations have been made to fill some gaps and are explained in each and every case, with the expectation that experts will propose better solutions. In some cases, administrative data have been used, knowing that they present biases due to their specific purpose. For example, administrative data on water withdrawals and returns of waste water are collected for the purpose of the payment of fees to water agencies. Biases are linked to the process of data collection, the existence of exemptions for small amounts (which in total can be significant in some places), and changes in water laws which modify these rules. These (important) data are made available by holders with appropriate warnings but, in the absence of other information on key features, they have been used as a temporary solution.

In many cases, data were produced from models with a low spatial resolution which do not meet the 1 ha granularity of the compilation of ENCA accounts for the Rhône River catchment. Several datasets are available with a resolution of 1 km², and even 18 km x 18 km in the case of the European Water Accounting (swbEWA). The ENCA data model in these cases consisted in resampling large grid-cells to 1 ha cells. This intermediate calculation produced values which were correct for a group of pixels of a size similar to the one used for the input data, but not for individual pixels. Thus, the values of the input grid are redistributed into various zonings. It is clear that the finer the resolution of input data, the better the resampled intermediate values.

Furthermore, accounts are expected to be produced on an annual basis. In the case of water, the intra-annual variability is important due to seasonal stress (in temperate Europe at least). In annual accounts, the stress resulting from this variability is taken into account, in particular for rivers and soil, as an index of stress (*see Table 4.1, section T4*). In the case of water accounts, swbEWA, a series of monthly datasets from January 2000 to December 2016, has constituted the basis for

¹⁴ *Combinaison multi-modèle et cartographie de consensus du débit de référence d'étiage et du débit moyen à l'échelle de la France* » (Marine Riffard, Vazken Andréassian, Pierre Nicolle et Julien Peschard, Cemagref/Irstea, UR HBAN, Antony, Avril 2012). Data are available at <http://www.onema.fr/debits-d-etiage-une-cartographie-nationale-de-consensus-0>

producing an annual time series of water accounts. Other datasets are annual (withdrawals and returns of waste water). Others are available for a couple of dates (WFD reporting) or represent one time points. In this latter case, data have been interpolated or extrapolated in order to reconstitute the full time series for 2000-2016 on the basis of the measured variability within the input meteorological data provided by swbEWA.

4.3.2 The data requirements of ENCA Water and Rivers

The proof-of-concept elaboration for water and river accounts in the Rhône River catchment follows the full set of detailed tables presented in the CBD TS77 ENCA-QSP (Weber, 2014). A few minor modifications have been done to classifications of uses to reflect what is available in current statistics. The main change consists in introducing a double rating of sustainable intensity of use.

The integrated tables below (**Table 4.1**) detail the set of ENCA-QSP parameters that are meaningful and more convenient for the purpose of the present assessment.

In the tables, the variables are presented in their context and indication is given of the specific data sources which have been employed. Variables which have not been addressed in the Rhône catchment ENCA test are also indicated. All the tables processed will be transferred on a DBMS, likely PostGIS database.

Table 4.1. Overview of data used for water accounts

T1. Ecosystem Water Basic Balance	
T1.A Ecosystem Water Basic Balance/ Stocks	
W1 [L]	Lakes & reservoirs
W1 [R]	Rivers & other streams
W1 [G]	Glaciers, snow & ice
W1 [N]	Groundwater
W1 [S]	Soil & Vegetation
W1	Opening Stocks
W4	Net Ecosystem Water Balance (NEWB) = W2-W3
W42	<i>Adjustment of NEWB = (W2-W3)-W4</i>
W5 [L]	Lakes & reservoirs
W5 [R]	Rivers & other streams
W5 [G]	Glaciers, snow & ice
W5 [N]	Groundwater
W5 [S]	Soil & Vegetation
W5	Closing Stocks = W1+W4+W42
T1.B Ecosystem Water Basic Balance/ Inflows	
W21	Precipitations
W221	<i>Surface runoff to rivers</i>
W222	<i>Infiltration/percolation</i>
W223	<i>Groundwater draining to rivers</i>
W224	<i>Groundwater draining to soil</i>
W225	<i>Other transfers received/ rivers (melting of glaciers)</i>
W226	<i>Others transfers received/ rivers (agriculture draining)</i>
W227	<i>Groundwater transfers received/ tables</i>
W228	<i>Other transfers received</i>
W22	Internal spontaneous water inflows [transfers received]

W231	Natural inflows from upstream territories
W232	<i>Intrusion of sea water</i>
W23	Natural inflows from external territories
W242	<i>Artificial inflows of water from other territories (distribution and transfers)</i>
W243	<i>Withdrawal of water from the sea</i>
W24	Artificial inflows of water from other territories and the sea
W251	<i>Returns/discharge of treated waste water</i>
W252	<i>Returns/discharge of untreated waste water/ used water</i>
W253	<i>Returns/discharge of untreated waste water/ urban runoff</i>
W25	Waste water returns/discharge to inland water assets
W261	<i>Losses of water in transport and storage</i>
W262	<i>Irrigation water</i>
W263	<i>Return of water from hydroelectricity production</i>
W264	<i>Return of mine water</i>
W265	<i>Return of water from cooling</i>
W266	<i>Return of water from other production</i>
W267	<i>Other returns of water</i>
W26	Other returns of abstracted water to inland water assets (incl. losses and irrigation)
W2	Total of water inputs and returns = Total increase of stocks of water

T1.C Ecosystem Water Basic Balance/ Outflows		
W311	<i>Spontaneous actual evapo-transpiration from rainfed agriculture & pasture</i>	
W312	<i>Spontaneous actual evapo-transpiration from forests</i>	
W313	<i>Spontaneous actual evapo-transpiration from natural land</i>	
W314	<i>Spontaneous actual evaporation from water bodies</i>	
W315	<i>Spontaneous actual evaporation from artificial land</i>	
W31	Spontaneous actual evapo-transpiration	
W321	<i>Surface runoff to rivers/ precipitations</i>	
W322	<i>Infiltration/ deep percolation</i>	
W323	<i>Groundwater draining to rivers</i>	
W324	<i>Other transfers provided/ rivers (melting of glaciers)</i>	
W325	<i>Others transfers provided/ rivers (agriculture draining)</i>	
W326	<i>Groundwater transfers provided/ tables</i>	
W327	<i>Others transfers provided/ soil (= W322)</i>	
W32	Internal spontaneous water outflows [transfers supplied]	
W331	<i>Natural outflows to downstream territories</i>	
W332	<i>Natural outflows to the sea / rivers</i>	
W333	<i>Natural outflows to the sea / groundwater</i>	
W33	Natural outflows to downstream territories and the sea	
W34_A&B	<i>Abstraction from surface water</i>	
W34_C	<i>Abstraction from groundwater</i>	
W34_F	<i>Abstraction from the sea</i>	
W34	Abstraction from water assets	
W351	<i>Collection of precipitation water (rainwater harvest)</i>	
W352	<i>Collection of urban surface runoff</i>	
W35	Abstraction/collection of precipitation water and urban runoff	
W36	Actual evapo-transpiration induced by irrigation	
W37	Evaporation from industry and other uses	
W381	<i>Artificial discharge of untreated wastewater to the sea</i>	
W382	<i>Other artificial outflow to other territory and the sea</i>	
W38	Artificial outflow of water to other territories and the sea	
W39	Other change in volume of stocks and adjustment (+ or -)	
W3	Total outflows of water = SUM(W34 to W39)	
T2. Accessible basic water resource surplus		DATA SOURCES
W1 [L]	Lakes & reservoirs/ Opening stock	Carthage "Hydrologie surfacique" dataset for areas. Volumes values for large lakes found in Wikipedia, default values for others.

W1 [R]	Rivers & other streams/ Opening stock	Calculated from the ENCAT_ARBO module and estimations of standard river kilometers. Mean kmcn value taken as provisional default value of stocks.
W1 [G]	Glaciers, snow & ice/ Opening stock	Corine land cover for areas 2000, 2006 and 2012. Measurements of depth and change on a few points. See methodology.
W1 [N]	Groundwater/ Opening stock	Renewable water estimated from deep percolation, draining to rivers and abstraction of groundwater.
W1 [S]	Soil & Vegetation/ Opening stock	Estimated from the ESDAC database of soil depth (down to 1 m) and stones contents combined with soil humidity. See methodology.
W1	Baseline accessible stock carried over from previous year	
W21	Precipitations	As T.1 W21
W22	Internal spontaneous water inflows [transfers received]	As T.1 W22
W23	Natural inflows from upstream territories	As T.1 W23
W2a	Total natural renewable water resources (TNWR) = W21+W22+W23	
W24	Artificial inflows of water from other territories and the sea	As T.1 W24
W25	Waste water returns/discharge to inland water assets	As T.1 W25
W26	Other returns of abstracted water to inland water assets	As T.1 W26
W2b	Total secondary water resources = W24+W25+W26	
W32	Internal spontaneous water outflows [transfers supplied]	As T.1 W32
W33	Natural outflows to downstream territories and the sea	As T.1 W33
W6A	Net water resource / Lakes & reservoirs	Estimated by default as 10% of stock
W6B	Net water resource / Rivers & other streams	Total inflows - irregular resource - reserved runoff - (grey water - reserved runoff, when >0)
W6C	Net water resource / Glaciers, snow & ice	not addressed in the test
W6D	Net water resource / Groundwater	Opening renewable stock + inflows - outflows
W6E	Net water resource / Soil & Vegetation	Net primary resource
W6	Net primary & secondary water resource = W2a+W2b-W32-W33	
W711	Irregular renewable water resources (-)	Rivers: Runoff exceeding the 3rd quartile of the period
W712	Restrictions of use to maintain environmental condition of rivers	W7121 or W7122
W7121	Legally reserved runoff (for dilution (BOD), aquatic life, navigation...) (-)	Reserved runoff = 2.5 % of mean annual discharge if > W7122, if not W7122
W7122	Grey water	= 40 times non treated wastewater
W713	Restrictions of use to maintain or restore groundwater	not addressed in the test
W714	Outflow secured through treaties, agreements, regulations or laws (-)	not addressed in the test
W715	Water natural resource unusable due to quality (incl. salinity) (-)	not addressed in the test
W716	Remote inaccessible water resources (-)	not addressed in the test
W717	Exploitable irregular renewable water resources/ annual storage (+)	not addressed in the test

W718	Previous net accumulation in water stocks (+ or -)	not addressed in the test
W719	Other accessibility adjustments of natural water (+ or -)	
W71	Total adjustment of natural renewable water resources (+ or -)	
W7a	Exploitable natural water resources = W2a+W71+W39	
W721	Secondary water resource unusable due to quality (-)	Untreated wastewater
W722	Other accessibility adjustments of secondary water (+ or -)	not addressed in the test
W72	Total adjustment of secondary renewable water resources	
W7b	Exploitable secondary water resources = W2b+W72	
W39	Other change in volume of stocks and adjustment (+ or -)	
W7	Net Ecosystem Accessible Water Surplus [NEAWS] = W7a+W7b+W39	

T3. Total water uses		DATA SOURCES
W341	Abstraction for distribution	From the database on water abstraction of the RMC Water Agency.
W342	Abstraction for own use by agriculture (incl. for irrigation)	From the database on water abstraction of the RMC Water Agency. Corresponds to only one part of water consumed by irrigation but shows up the importance of groundwater used.
W343	Abstraction for own use for hydroelectricity production	From the database on water abstraction of the RMC Water Agency.
W344	Abstraction for own use by other production (incl. cooling)	From the database on water abstraction of the RMC Water Agency.
W345	Abstraction for own use by other production activity	From the database on water abstraction of the RMC Water Agency.
W346	Abstraction by municipal bodies [drink water]	From the database on water abstraction of the RMC Water Agency.
W347	Other abstraction by municipal bodies and households [drink water]	Estimation of self account abstraction not recorded above (municipalities and households only)
W81	Abstraction from water assets (W81 = W34)	From the database on Water abstraction of the RMC Water Agency. See methodology
W311	Spontaneous actual evapo-transpiration from rainfed agriculture & pasture [green water]	ETA on CLC Agriculture classes (in absence maps of irrigation, ETA models don't record it)
W312	Spontaneous actual evapo-transpiration from forests [green water]	ETA on CLC Forest classes
W82	Agriculture and forestry 'green water' use = W311+W312	
W831	Collection of precipitation water (rainwater harvest) = W351	not addressed in the test
W832	Collection of urban surface runoff = W352	= W352
W83	Abstraction/collection of precipitation water and urban runoff	
W8	Total Use of Ecosystem Water [TUEW]	
W911	Use of water received from distribution/ Communes and Households	Called Drink Water (Eau potable) in the RMC Water Abstraction datasets
W912	Use of water received from distribution/ Agriculture (incl. Irrigation)	Distributed irrigation water is poorly taken which means that consumption of irrigation water is underestimated.

W913	<i>Other use of water received from distribution</i>	<i>Recorded in RMC Water Abstraction datasets. See methodology</i>
W914	<i>Wastewater received for treatment</i>	<i>Recorded in RMC Wastewater Treatment plants datasets. See methodology</i>
W91	Use of water received from other economic units [distribution and treatment]	
W921	<i>Artificial transfers of water for distribution (incl. municipal and irrigation water)</i>	<i>not addressed in the test</i>
W922	<i>Artificial transfers of water for hydroelectricity</i>	<i>not addressed in the test</i>
W923	<i>Artificial transfers of water for cooling</i>	<i>not addressed in the test</i>
W924	<i>Other artificial transfers</i>	<i>not addressed in the test</i>
W92	Artificial transfers of water from other territories (W92=W242)	
W93	Withdrawal of water from the sea (W93=W242)	
W94	Re-use water within economic units	not addressed in the test
W95	Imports of Water/ commodities & residuals content	not addressed in the test
W96	Exports of Water/ commodities & residuals content	not addressed in the test
W9	Direct Use of Water = W8+W91+W92+W93+W94+W95	not addressed in the test
W10	Domestic Consumption of Water = W9-W96	not addressed in the test
W11	Virtual water embedded into imported commodities	not addressed in the test
W12	Total Water Requirement = W9+W11	not addressed in the test

T4. Table of indexes of intensity of use and ecosystem health		DATA SOURCES
W7	Net Ecosystem Accessible Water Surplus [NEAWS = W7a+W7b]	
W8	Total Use of Ecosystem Water [TUEW]	
W131	Intensity of water use = W7/W8	
W132 [L]	Quantitative status index/ Rivers and other water streams	not addressed in the test
W132 [R]	Quantitative status index/ Lakes & reservoirs	not addressed in the test
W132 [G]	Quantitative status index/ Snow & glaciers	not addressed in the test
W132 [N]	Quantitative status index/ Renewable groundwater	EU WFD reporting / EauFrance
W132 [S]	Quantitative status index/ Soil	Assessed from monthly data on soil humidity (sbwEWA)
W132	Composite index of water resource quantitative status	
W13 [SIWU]	Sustainable intensity of water use = AVG(W131*W132), [0,1]	
W141 [L]	Bio-chemical status/ Rivers and other water streams	not addressed in the test
W141 [R]	Bio-chemical status/ Lakes & reservoirs	not addressed in the test
W141 [G]	Bio-chemical status/ Snow & glaciers	not addressed in the test
W141 [N]	Bio-chemical status/ Renewable groundwater	WFD reporting / EauFrance
W141 [S]	Bio-chemical status/ Soil	not addressed in the test
W142	Vulnerability to pollution	Zones of vulnerability to nitrates (reporting to EU Nitrates Directive/ EauFrance)
W143	Human diseases caused by ecosystem water	not addressed in the test
W144	Animal and plant diseases caused by ecosystem water	not addressed in the test
W145	Impacts of water management on ecosystem health/ agriculture draining	quickly addressed in the test from data on draining by municipalities (areas) and default values for volumes
W146	Dependency from artificial inflows	not addressed in the test; could be calculated from T3. Water Use Account
W14x	Other...	not addressed in the test
W14 [CIEH]	Composite Index of Ecosystem Health [0,1]	
W15 (EIUV)	Water Ecological Internal Unit Value = AVG(W13 [ISUE]*W14 [ICES])	

4.4 Methods

Ecosystem water accounts are integrated by ENCAT and downscaled to UZHYD and SELU. The main lines of the methodology used for this purpose is listed in **Supplementary File 4.1** to inform on the complete set of procedures and variables that constitute the ecosystem water accounts. Access to the complete methodology of water accounting is provided through the link². We detail one example to illustrate the computation of rivers srkm values for compiling rivers quality accounts.

Computation of rivers srkm values for compiling rivers quality account

This is the chosen example because it highlights the link of water and rivers account. It has been established on the basis of the calculation of “standard river kilometres” (srkm) carried out in the test. However, in the course of the exercise, it has appeared that the solution proposed in the French “Comptes du Patrimoine Naturel” of 1986 and retained in the SEEA-Water was focused on water quality. The weighting factor for rivers’ length, the discharge measured in m³/second is correct in this case and is used for producing the *water quality index*. Concerning the *rivers ecological condition*, the conventional weighting system obviously overweighs the large rivers and in particular the main drain. Medium-size rivers may have similar ecological importance and should not be discriminated according to their runoff. For this reason, a second type of srkm has been defined for accounting for rivers ecological status where the natural logarithm of discharge is used.

Principles of river water quality accounting

SKRM values have been defined as to integrate quantitative and qualitative accounts, weighting factor allowing comparisons of rivers of various sizes.

Accounting for the quality or ecological status of rivers involves two elements:

- (1) a scale (s) of values such as the one (s) produced for reporting to the EU Water Framework Directive (WFD) on the good ecological and chemical status of river water bodies;
- (2) a reference population composed of clearly defined and commensurable entities. In the case of rivers, the differences between the average flows of large rivers and small rivers easily vary from 1 to 1000 or more, which makes the weighting of lengths by notes uncertain and does not allow significant aggregations. This question is well known to the statisticians and led the French Natural

Patrimony Accounts of 1986 to propose a solution based on the Haldal and Østdal method of sampling “populations” of river stretches for the Telemark basin in Norway¹⁵.

This methodology consists in weighting the river sections by the product of their length by their flow, the unit being called standard river-kilometre (SRKM). It is important to note that SRKMs are additive. This metric is common to all rivers and one can aggregate the statistical results for all rivers, or a category of them (large, small ...) and make significant comparisons. The srkm method was incorporated to the 2009 SEEA-Water. ENCA follows this methodology.

Quality of water and condition of rivers: two different SRKM units

SRKM have been used to measure the quantity of running water in order to measure rivers water quality with in mind its relation to water use and pollution. As long as water quality is the objective, it is normal that the weighting factor relates to the quantity of water, measured by the discharge. This is the approach followed for water accounts. Let us call it **srkm_W** (W for Water). For river systems condition accounts, a different way of weighting is used called **srkm_R** (R for River), where discharge is replaced by its natural logarithm (ln discharge).

The argumentation is as follows. Considering river systems condition in the context of ecosystem accounting, the orders of magnitude are different from those of water use. Water runoff is certainly important, but not exclusively. Two rivers with the same length but with different discharge rates, let us say in the proportion of 1 to 2 or 3 will have the same importance considering the ecosystem landscape. Other dimensions than the mere quantity of water will import, evenly adjusted by the quality of water: regularity of the flow, support to aquatic life, support to the neighbouring environment, etc. It is difficult to argue that 1 km of river with a discharge rate of 1000 m³/s is equivalent (has the same value) to 1000 km of river at 1 m³/s. In terms of water resource, it is equivalent, not in terms of potential for the ecosystem. Due to the fact that water flows cumulate from source to estuary, conventional srkm are a correct methodology for water resource and integrating volumes and water quality. It is a good measurement of rivers as assets from this point of view and it is used in the water account. Regarding river systems and their function in landscape systems, the conventional weighting system will hugely overweight the downstream part of the

15 Haldal, Johan and Østdahl, Torbjørn, 1984, *Synoptic monitoring of water quality and water resources. A suggestion on population and sampling approaches*, Statistical Journal of the United Nations Economic Commission for Europe, vol. 2, no. 4, pp. 393-406, 1984, DOI: 10.3233/SJU-1984-2406

largest rivers. It is therefore proposed for river systems accounting in ENCA to adopt a second definition of srkm, giving a less excessive weight to the water flow. The methodology of multiplying rivers lengths by discharge is the same, but the gross value of discharge is expressed logarithmically (ln). Large rivers will loose some weight: the ratio of 10 to 100 becomes 2.3 to 4.6, and the ration of 10 to 1000 becomes 2.3 to 6.9, etc. We call **srkm_R** the new standard unit defined as $1 \text{ km} \times \ln(1\text{m}^3/\text{s})$.

Methodological considerations for implementation

While srkm_W is used in the water account for integrating water quality, srkm_R is used in the Ecosystem infrastructure integrity account, the sub-account of river systems. It allows computing NREP, the Net River Ecosystem Potential (see **Section 4.5 and Chapter 5**), a component of NTEP, the Net Total Ecosystem Potential.

The implementation of basin-wide river km calculation (srkm_W and srkm_R in the same run) requires a large amount of flow data. It was one of the EEA ECRINS project initial ambition to allow this calculation following the experiences of water quality accounts making use of srkm in France, Spain, England, Ireland, and Slovenia. It has not yet been achieved because of insufficient quality of the rivers geometry in ECRINS (JRC's CCM2 map produced with a digital elevation model). It has generated modelling issues, and as an ancillary consequence, the use of ECRINS's data once foreseen was abandoned for the Rhône project.

In France, it would have been possible and rather straightforward to implement the SRKM methodology using Carthage and "map of consensus" of average rivers runoff produced by IRSTEA for ONEMA¹⁶. The existence of the IRSTEA dataset was not known when the issue of computing rivers srkm values was addressed, and a second best solution was attempted.

Indeed, the difficulty of implementing srkm had been perceived in France by the IFEN and the answer has been a simplified calculation method where a pseudo-hierarchy of classes of streams treated statistically replaces the topological description. This method and first quality water accounts have been published by IFEN¹⁷ in 1999.

¹⁶ In the case of Switzerland, the "natural flows" associated with the river sections of the various classes of the FOEN database should allow easy calculation, but have not been tested.

¹⁷ Crouzet, P., Germain, C. et Le Gall, G., 1999, Les Comptes de la qualité des cours d'eau, Mise en œuvre d'une méthode simplifiée de calcul, Études et travaux no 5, IFEN/ OFEV, Orléans.

The IFEN simplified method was based on river statistics for some twenty watersheds. It was implemented quickly with less efficient computational means and much less data than those available today. For the ENCA Rhône water accounts project, this methodology has been reviewed and adapted with the available geographical data on rivers and hydrological units.

Implementation of the simplified method for calculating kmcn in the context of ENCA

It is important to keep in mind that SRKM measurements concern stocks (length weighted by mean flows) and not flows of water. The flow account is established in full by ENCAT only. The calculation of srkm is done by unit hydrological zones (UZHYD) to account for the grouping of small rivers and the variability of hydro-geographical conditions within ENCATs. However, remember UZHYDs are not interconnected by the ARBO tree coding.

The first work consisted in calculating with the module ENCAT_ARBO the discharge of ENCAT main drains. These were multiplied by the corresponding lengths of rivers main drains to have their value in SRKM.

In a dozen of cases where secondary drains (some class 2 and 3 rivers) run through several UZHYDs (see above), their flows were estimated in the same way as for the main drains in reference to the total area of the ENCAT and not that of the various UZHYD they cross. Their "average" throughput by ENCAT is then multiplied by their length in each UZHYD. The results by UZHYD are finally grouped by ENCAT (**Figure 4.13**). The figure illustrates the cartography of the river potential stocks at UZHYD and ENCAT scales.

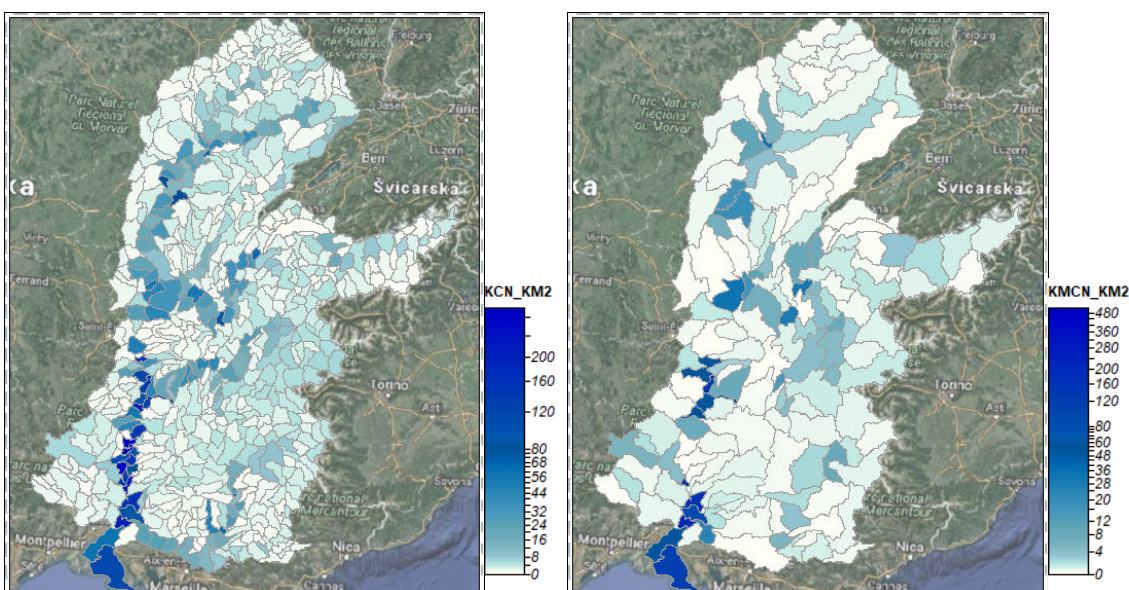


Figure 4.13. Rivers potential stocks in SRKM/km² simplified methodology, by UZHYD (left) and aggregated by ENCAT (right). Dark color correspond to high potential values.

4.5 Results - Rivers quality/ status account

It is understood here that the chemical quality is that of water, while ecological status relates to rivers systems condition.

4.5.1 Contextualizing the work

The methodology of accounting for river water quality has been defined by Natural Patrimony Accounts in France and adopted and developed for the water accounts in Spain¹⁸. It was tested at the European Environment Agency and later on adopted by the SEEA-Water in 2008. It consists of defining a reference population of homogeneous river stretches, calculate for each of them a value in srkm (quantitative measure) and multiply it by a status index. For rivers condition account, the unit used is **srkm_R**. The values in srkm_R of the different river classes are calculated at UZHYD scale in the first place (see below).

We estimated that the most appropriate data available for river quality accounts are those of the reporting to the European Water Framework Directive 2009 and 2015 (WFR), in the first place that on the "good ecological status" of the river water bodies. It has the benefit of being very inclusive, produced by experts and to provide a rating scale that can be translated into rating indexes. The WFR reporting data on the chemical quality of rivers could also be used, but they have the disadvantage of being poorly nuanced ("Good" and "Not in good condition"). The WFR chemical indicators are used in the table of intensity of use and health indexes of the water accounts, not for the river systems accounts.

The analysis of physico-chemical and biological monitoring variables has been rejected at this stage because it goes beyond accountants' competence. It is clear, however, that if summary maps of previous years are available and consistent with the current report, their use for historical ecosystem accounting of water should be considered. One can think in particular of the syntheses produced in the framework of the SEQ-WATER (System of evaluation of the quality of the water), but also data of the "historical" inventories which have been used for the first tests with the *Institut français de l'environnement* (IFEN). The intrinsic comparability of these various inventories is probably not very high, but it must be borne in mind that ecosystem accounting is primarily about measuring

18 Exergo-ecological approach developed by Jose Manual Naredo and Antonio Valero. See Valero A., et. alii, 2009, Physical Hydromonics: Application of the exergy analysis to the assessment of environmental costs of water bodies. The case of the inland basins of Catalonia, Energy, Volume 34, Issue 12, December 2009, Pages 2101-2107 <https://doi.org/10.1016/j.energy.2008.08.020>

changes across good to bad quality classes. Assessing broad trends in that context would be meaningful even though observation methods have evolved.

4.5.2. On the “Ecological Status”

The theoretical model for calculating the ecological status is presented in the **Figure 4.14**.

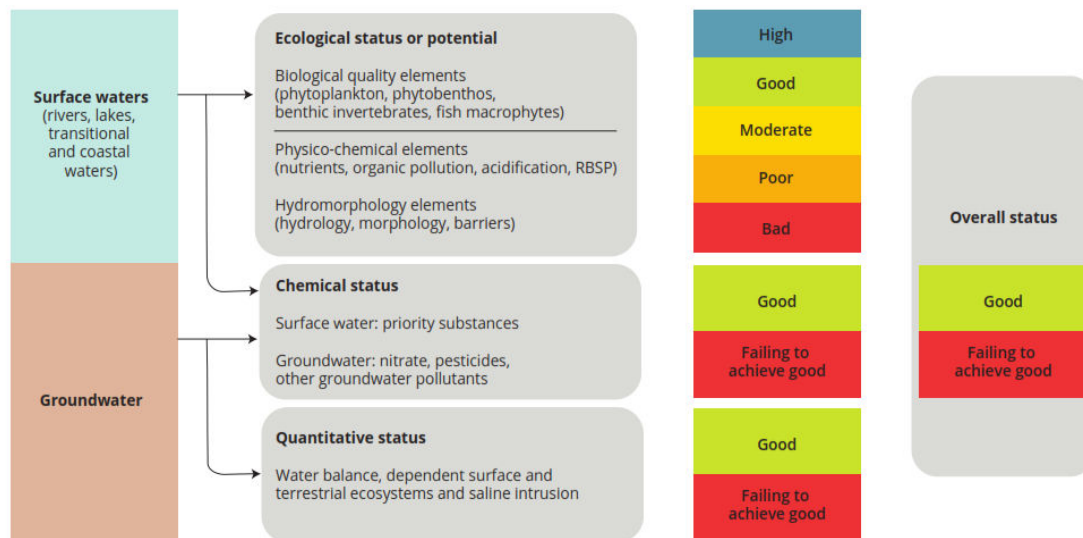


Figure 4.14: Assessment of status of surface water and groundwater according to the WFD. EEA report on European waters / Assessment of status and pressures 2018¹⁹

We quote “Ecological status is based on biological quality elements and supporting physico-chemical and hydromorphological quality elements.” (EEA, 2018, page 7), and “Physico-chemical quality elements support the biological quality elements. They generally consist of (1) light and thermal conditions, (2) oxygenation conditions, (3) salinity, (4) nutrient conditions, and (5) river basin specific pollutants (RBSPs). In rivers and lakes, they also include acidification condition.” (EEA, 2018, idem)

The ecological status of rivers analysed here is in agreement with the reporting to the EU Water Framework Directive. Our synthesis is in adequacy with what the water and the river accounts of the ENCA attempt to describe.

Regarding the measurement of the hydromorphological quality that is considered responsible for the failure to reach the good ecological status of 40% of the water bodies in Europe, an important work

¹⁹ EEA, 2018, European waters / Assessment of status and pressures 2018, EEA Report No 7/2018
<https://www.eea.europa.eu/publications/state-of-water>

is carried out in France in the frame of ARHYCE²⁰. In the reports to the WFD, *"the importance of the rupture of the ecological continuity appears as the major degradation (11.5% of the degradations)"*. The methodology for measuring river fragmentation is not fully explained in the documents consulted and it is not possible to decide whether this variable is taken into account at the same level as for terrestrial ecosystems. However, it seems preferable to consider that the report on the ecological status of French rivers integrates this dimension sufficiently and not undertake at this stage fragmentation accounts of the rivers from the available database on obstacles to river runoff.

4.5.3 Ecological Status data

2009 and 2015 data on the ecological status of water streams/ water bodies have been downloaded from Cartograph (**Figure 4.14**) <http://www.cartograph.eaufrance.fr/donnees/203571/2015>. It should be noted that only the attribute tables with the WFD water bodies' codes can be downloaded, not the vector mapping files (e.g. .shp).

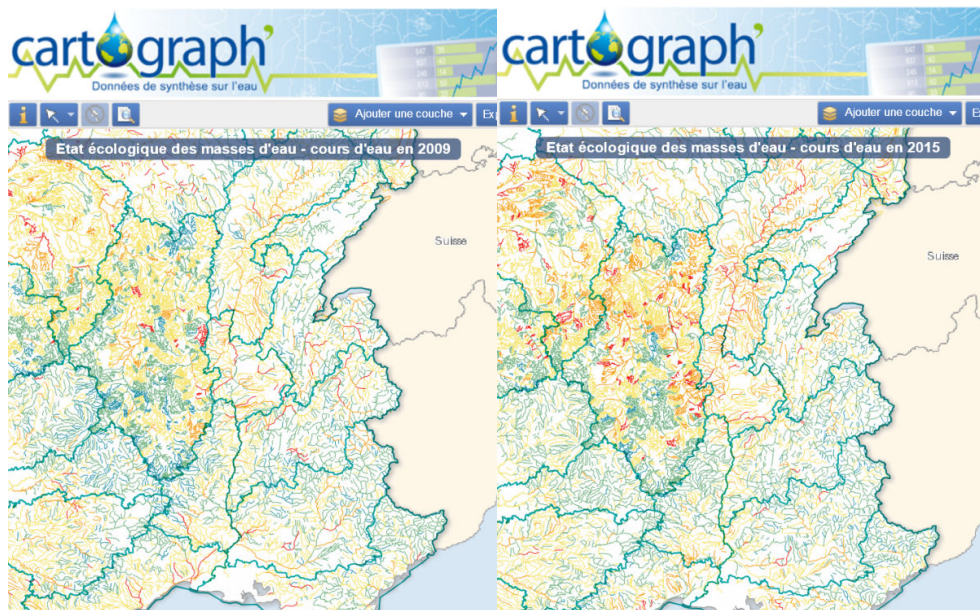


Figure 4.15. Data source on water streams ecological status in 2009 (left) and 2015 (right).

Definition of an ecological status rating scale

²⁰Gob, F., Bilodeau, C. et al. *Un outil de caractérisation hydromorphologique des cours d'eau pour l'application de la DCE en France (CARHYCE)* <https://journals.openedition.org/geomorphologie/10497>

After several sensitivity tests, the five state classes were converted for application into notes according to the rule that the very good quality has a score of 1, a downgrading of a class results in a relative loss of 25%. This leads, after rounding, to the following scoring grid:

Very good = 1, Good = 0.8, Average = 0.6, Poor = 0.4 and Bad = 0.3.

Cartograph tables in .csv format for 2009 and 2015 were merged and the notes introduced according to the scale described above (**Table 4.2**). More river water bodies have been reported in 2015 than in 2009. In order to ensure better coverage of the account, in case of missing data in 2009, the values of 2015 were conventionally adopted. The file is named EtaEco_FRDR_MDORIV_09_15.csv

Table 4.2. Extract from the file EtaEco_FRDR_MDORIV_09_15.csv

DCEMDO	LIBELLE	Etat 2009	QL09	Etat2015	QL15
FRDR10001	RIVIERE LA FRAYERE D'AURIBEAU	Moyen	0.6	Moyen	0.6
FRDR10002	RUISSEAU DE REVILLON	Bon	0.8	Mediocre	0.4
FRDR10003	RUISSEAU LE SONNANT D'URIAGE	Moyen	0.6	Mediocre	0.4
FRDR10004	AUBANEDE	Bon	0.8	Mediocre	0.4
FRDR10005	RUISSEAU DE CHARSAC	Bon	0.8	Bon	0.8
FRDR10006	TORRENT DU TOUROND	Bon	0.8	Tres bon	1
FRDR10007	TORRENT DU LOMBARD	Bon	0.8	Tres bon	1
FRDR10008	TORRENT DU MELEZET	Tres bon	1	Bon	0.8
FRDR10009	RUISSEAU LA BRETTE	Bon	0.8	Bon	0.8
FRDR10010	RUISSEAU LE VEZY	Bon	0.8	Moyen	0.6
FRDR10011	RUISSEAU D'ANTERNE	Tres bon	1	Tres bon	1
FRDR10012	TORRENT DE DURMILLOUSE	Bon	0.8	Bon	0.8
FRDR10014	TORRENT DE BLEME	Bon	0.8	Bon	0.8

4.5.4. Extraction of rivers status data to hydrological units (UZHYD)

Water bodies are coded according to the codification of the Water Framework Directive (DCEMDO), and do not include a HYDRO code or BD Carthage river class identifiers used for accounting. The following procedure was used:

- (1) A join has been done between the river files, using GIS command;
- (2) The shape file of rivers (lines) was intersected with the shapefile of UZHYD (polygons). This stage is necessary for Class 1, 2 or 3 rivers that cross several UZHYDs and for which ecological status information is only available for long or very long sections. It was performed with SAGA

Chapter 4 Ecosystem Water Account

Module Shapes / Lines / Line-Polygon intersection. After cleansing from useless items, the file was saved as QUAL_RIV_CALC.shp;

(3) Ecological status notes were assigned to river stretches of QUAL_RIV_CALC.shp, using GIS “join” module. Manual checks and corrections were necessary because the water body codifications for the WFD are not fully consistent. The problem is essentially the Rhône *per se* which has a simplified code in some cases, complete in others. In addition, corrections had to be done for the Rhône downstream of *Beaucaire* and the *Petit Rhône* as they are considered in WDF as “transitional water”, while ENCA considers them as rivers. Thus, values given for both were: QL09 = 0.8 and QL15 = 0.6.

4.5.5 Calculation of average ecological status values by river class and by UZHYD

SQL queries were done to calculate the average ecological status values by river class and by UZHYD with SQLite Studio (but can be carried out with any DBMS).

The typical query for one river class 1 to 6 reads:

```
SELECT CD_UZHYD, AVG(QL09), AVG(QL15)
FROM QUALRCALC
WHERE CL_RIV = 1
GROUP BY CD_ZHYD
```

Subsequently, we selected CD_UZHYD (hydrological units code) and the average of the QL09 and QL15 values (quality index, from the QUALRCALC) in order to extract values for river stretches of class one and group results by hydrological units.

The last step consisted in multiplying for each river class (C1 to C6) of each UZHYD the AVG(QL) index by river values in srkm in order to obtain Quality weighted srkm (QL KC). In the spreadsheet or database, the fields involved are shown in **Figure 4.16**.

RIV1QL09	RIV2QL09	RIV3QL09	RIV4QL09	RIV5QL09	RIV6QL09
RIV1QL15	RIV2QL15	RIV3QL15	RIV4QL15	RIV5QL15	RIV6QL15
×					
KCNMOY C1	KCNMOY C2	KCNMOYC 3	KCNMOYC 4	KCNMOY C5	KCNMOYC 6
=					
QLKC09C1	QLKC09C2	QLKC09C3	QLKC09C4	QLKC09C5	QLKC09C6
QLK15C1	QLK15C2	QLK15C3	QLK15C4	QLK15C5	QLK15C6

Figure 4.16. Principle of calculating quality weighted srkm by hydrological units

This methodology needs to be modified to correct a bias in WFD reporting because circa 20% of rivers are not reported. Most if not all rivers of class 1 to 3 and even 4 are fully recorded, but a number of rivers of class 5 are missing and very few rivers of class 6/7 are present in the dataset. This means that the above calculation gives the latter a weighting of 0. This is all the more regrettable since in most cases small rivers are of better quality than larger ones. In addition, the quality accounts methodology seeks to account for all rivers, while the usual quality statistics only cover large ones. In order to minimize the issue, “no values” have been replaced by 1 in the initial quality indexes table.

Methodological considerations

The calculation of average quality scores by UZHYD does not take into account rivers weight in srkm. It is to some extent in contradiction with the principle of the weighting river reaches per srkm, which is the basis of the accounts. However, its application by UZHYD for each river class taken separately limits the bias since river classes are relatively homogeneous.

In the case of no quality values (very small rivers), the average quality observed by UZHYD could have been retained instead of the value 1. However, this choice would avoid bias only in the case of a stretch of large river of poor quality belonging to an upstream UZHYD. Small rivers with no data would have in this case an arbitrary note. In consequence, giving 1 is a better solution in a majority of cases.

First results of the ecological status of rivers accounts

We have generated results at three spatial hydrological scales (UZHYD, ENCAT, and Sub-basins) with the aim of addressing and evaluating the following major parameters.

1. **River Water Potential** calculated based on $\text{length} \times \text{discharge} = \text{SRKM_W}$ and its derivative river potential stock which is the SRKM_W by surface unit.
2. **River Condition Potential** calculated as $\text{SRKM_R} \times \text{ecological status index}$. Synonyms of this term appear in the literature (SEEA-EEA and WFD documents): “River ecological condition” or “River quality potential”.

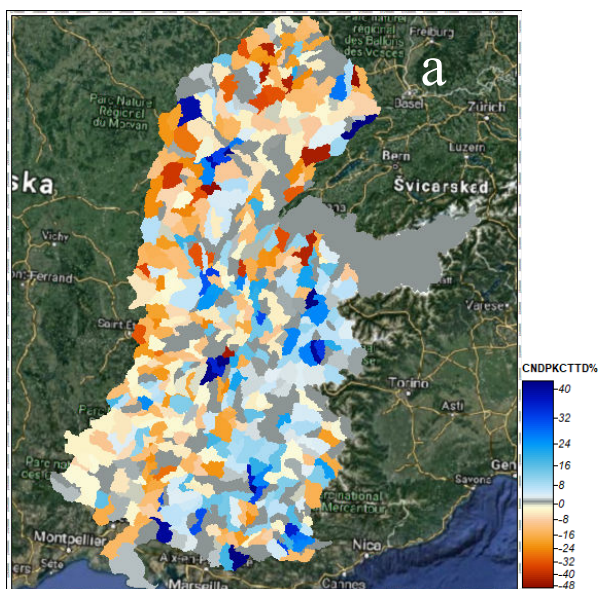
In addition, the use of the river weighting method per srkm of allows for relevant aggregations and comparisons:

In space: between basins, administrative units, and other zonings

In time

Between classes of rivers.

The following maps show very marked changes for a relatively short period of 6 years. Results by UZHYD are presented in **Figure 4.17**.



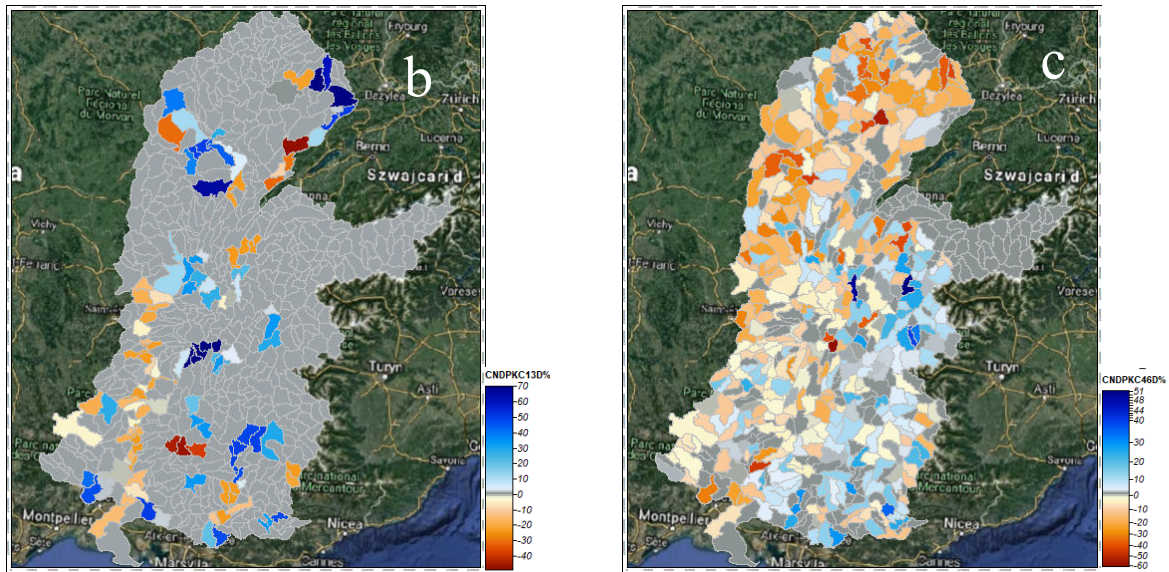
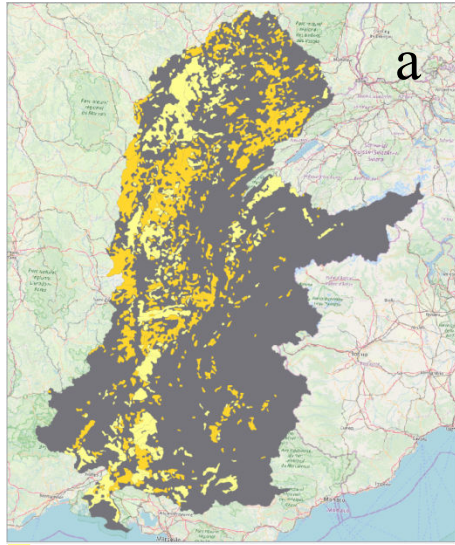


Figure 4.17. Change in ecological status by UZHYD (weighted $srkm_R$, %) 2009-2015, All rivers (a); Large rivers (classes 1 to 3) (b); and Small rivers (classes 4 to 6) (c). Cool colors represent high values.

The values of the large rivers cumulate for each UZHYD the sum of local values and the upstream contributions. One can see that the change in ecological status along the Rhône axis is very marked (panel b). Overall, the all rivers situation bears essentially the footprint of small river pattern.

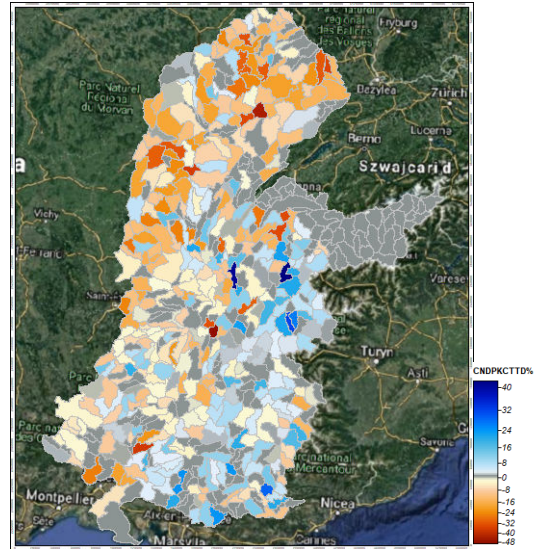
Concerning the small rivers, sources of pollutions are mainly internal to UZHYD. The map in panel c shows a marked deterioration on the western part of the basin, especially in the plain and hillside areas, probably related to agriculture (Figure 4.18). Mountain areas seem less affected or their state has even improved. Importantly, the contrast between the river water potential stock (Figure 4.12, left panel) and the change in ecological status (Figure 4.18, panel b) is noticeable.

These preliminary correlations will be further examined through the subsequent integration of water and river accounts with those of biomass (soils, agricultural production, and forests) and terrestrial ecological infrastructures and biodiversity (see Chapter 5).



20 Broad pattern agriculture
30 Mixed agriculture & pastures

Dominant Land Cover Types



Change in ecological status, (weighted srkm, %) 2009-2015, small rivers (classes 4 to 6), by UZHYD

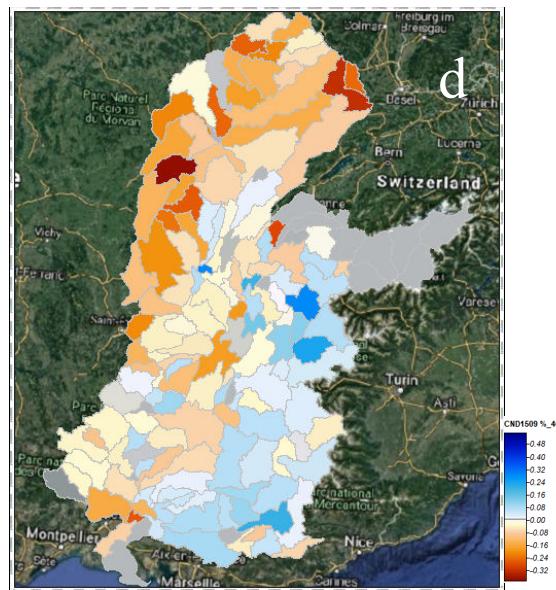
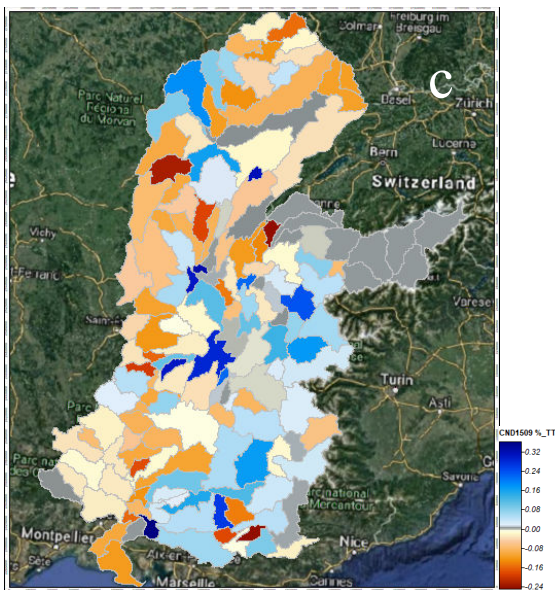


Figure 4.18. Visual comparison between spatial distribution of agriculture panel (a) and changes of small rivers ecological status panel (b). Change in ecological status (weighted srkm_R, %) 2009-2015, All rivers by ENCAT, panel (c), and Small rivers (classes 4 to 6) by ENCAT, panel (d). Hot colors in panels b, c, and d indicate degradation. As with the UZHYD results from above, the ENCAT small rivers pattern captures the same trends in the change of the ecological status.

Preliminary results by sub-basins: maps, accounts, and graphs

The ENCA sub-basins (SB_CECN or SB_ENCA) were defined according to the tree coding of ENCATs (see Section 4.2.2). Their River Water Potential and River Condition Potential can be aggregated according to the srkm methodology (Figure 4.19).

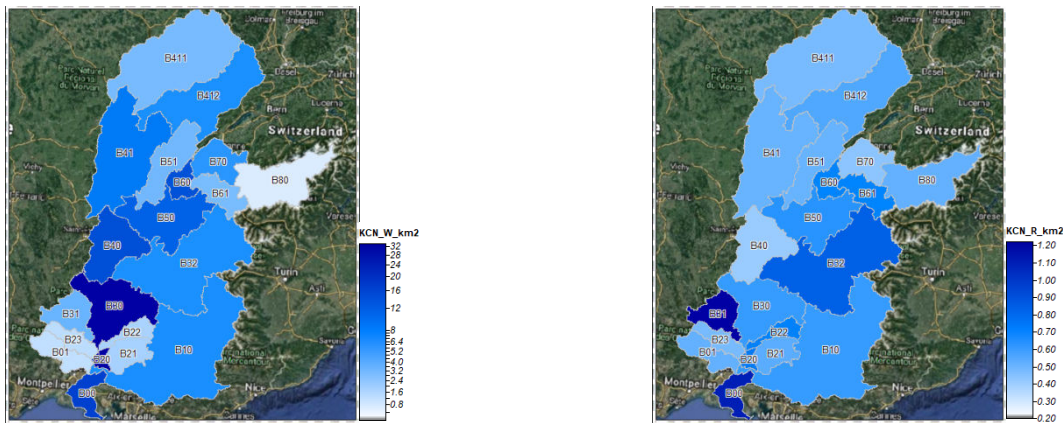


Figure 4.19 River Water Potential and River Condition Potential at SB_ENCA scale. River Water Potential measured in SRKM_River/km² (left); River Condition Potential measured in SRKM_Water/km² (right).

Rivers Condition Potential is based on the srkm_Rivers unit. Using the srkm_Water unit weighted with water discharge rates would have given an overwhelming importance to the Rhône river and would have resulted in a picture of change in river condition difficult to understand. Using the srkm_Rivers unit allows producing a synthetic view of degradation or improvement trends by sub-basins, highlighting the respective contribution of large and small rivers to the processes. The assessment by sub-basins is the aggregation of assessments carried out at the level of elementary hydrological units for 6 different classes of rivers.

How the River Condition Potential is reflected in changes of the ecological status according to river classes is shown in Figure 4.20.

Chapter 4 Ecosystem Water Account

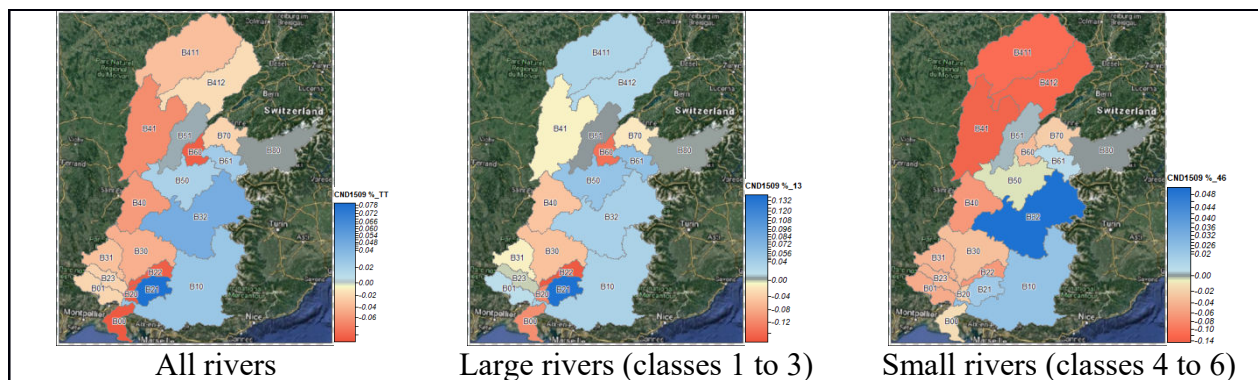


Figure 4.20. Change in Rivers Condition Potential (weighted srkm_R, %) 2009-2015, All rivers, Large rivers and Small rivers by SB_ENCA.

The maps illustrate results detailed in accounting tables and the corresponding charts of Rivers Condition Potential (see below), where quality is derived from WFD “ecological status” reporting.

Table 4.3. SB_ENCA River Condition Potential 2009.

units: srkm_R weighted by ecological status index

ENCA Sub-Catchments/ Sous-bassins CECN	River Condition Potential 2009						Total Rivers
	Rivers class 1	Rivers class 2	Rivers class 3	Rivers class 4	Rivers class 5	Rivers class 6	
SBCECN_00	580	749	0	13	25	38	1405
SBCECN_01	174	64	89	92	55	307	783
SBCECN_10	1183	913	343	867	931	2285	6522
SBCECN_20	179	0	0	10	5	17	211
SBCECN_21	0	267	89	56	104	194	710
SBCECN_22	192	0	22	62	134	238	648
SBCECN_23	180	0	48	57	61	145	491
SBCECN_30	987	200	179	352	326	552	2596
SBCECN_31	275	164	183	391	321	1186	2520
SBCECN_32	1403	175	790	834	1147	3362	7710
SBCECN_40	285	237	158	168	190	212	1250
SBCECN_41	354	933	180	543	345	387	2742
SBCECN_411	845	784	393	543	370	448	3382
SBCECN_412	1341	87	297	279	196	230	2431
SBCECN_50	597	380	82	383	391	683	2517
SBCECN_51	379	297	135	165	131	158	1265
SBCECN_60	269	0	143	42	77	165	696
SBCECN_61	228	0	116	143	144	420	1051
SBCECN_70	21	97	31	226	136	438	947
SBCECN_80	662	0	121	210	398	1414	2805
Total Rhône River Catchment	10134	5349	3398	5434	5487	12878	42681

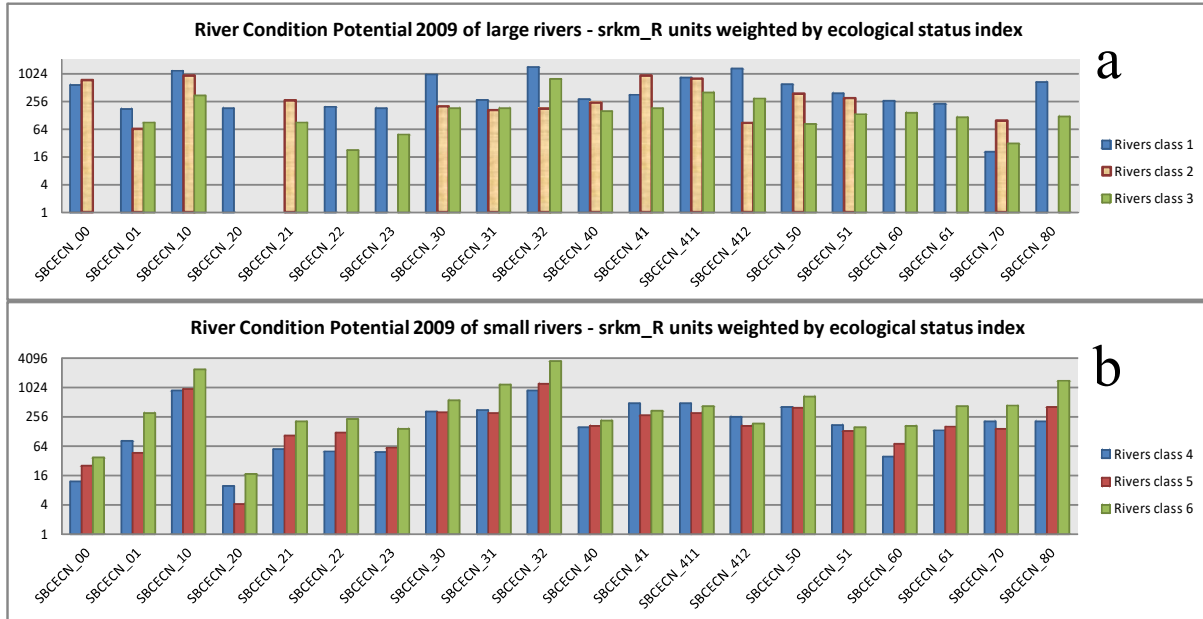


Figure 4.21. SB_ENCA River Condition Potential 2009, Large (a) and Small (b) Rivers.

Table 4.4. SB_ENCA River Condition Potential 2015.

units: srkm_R weighted by ecological status index

ENCA Sub-Catchments/ Sous-bassins CECN	River Condition Potential 2015						Total Rivers
	Rivers class 1	Rivers class 2	Rivers class 3	Rivers class 4	Rivers class 5	Rivers class 6	
SBCECN_00	496	669	0	12	25	38	1239
SBCECN_01	186	64	83	81	46	303	764
SBCECN_10	1249	943	334	872	955	2353	6705
SBCECN_20	156	0	0	10	4	17	187
SBCECN_21	0	312	94	55	105	201	767
SBCECN_22	142	0	29	50	122	231	574
SBCECN_23	178	0	48	48	58	145	478
SBCECN_30	900	208	175	324	312	548	2468
SBCECN_31	274	156	183	348	301	1163	2424
SBCECN_32	1532	158	771	881	1228	3497	8068
SBCECN_40	233	247	161	155	165	209	1170
SBCECN_41	354	918	174	485	270	334	2535
SBCECN_411	836	857	392	476	303	411	3275
SBCECN_412	1399	94	283	256	168	186	2385
SBCECN_50	664	382	73	396	383	669	2567
SBCECN_51	393	288	132	169	130	155	1266
SBCECN_60	202	0	143	39	71	163	618
SBCECN_61	228	0	136	131	158	423	1076
SBCECN_70	18	97	31	204	141	433	923
SBCECN_80	662	0	121	208	397	1414	2802
Total Rhône River Catchment	10102	5392	3364	5199	5342	12893	42292

Table 4.5. SB_ENCA Gains or Losses of River Condition Potential 2009-2015.

units: srkm_R weighted by ecological status index

ENCA Sub-Catchments/ Sous-bassins CECN	Gains or Losses of River Condition Potential 2009-2015						Total Rivers
	Rivers classe 1	Rivers classe 2	Rivers classe 3	Rivers classe 4	Rivers classe 5	Rivers classe 6	
SBCECN_00	-84	-80	0	-1	0	0	-165
SBCECN_01	12	0	-6	-12	-9	-4	-19
SBCECN_10	66	30	-9	5	24	68	184
SBCECN_20	-23	0	0	0	0	0	-24
SBCECN_21	0	44	5	-1	0	7	56
SBCECN_22	-50	0	7	-12	-12	-7	-73
SBCECN_23	-1	0	0	-9	-3	0	-13
SBCECN_30	-87	8	-4	-28	-13	-4	-128
SBCECN_31	-1	-8	0	-43	-20	-23	-96
SBCECN_32	128	-17	-18	48	81	136	358
SBCECN_40	-52	10	3	-13	-25	-3	-80
SBCECN_41	0	-15	-6	-58	-75	-53	-207
SBCECN_411	-9	72	0	-67	-67	-37	-107
SBCECN_412	58	7	-14	-23	-29	-44	-46
SBCECN_50	67	1	-9	13	-8	-14	50
SBCECN_51	14	-10	-3	4	-1	-3	1
SBCECN_60	-67	0	0	-3	-6	-2	-79
SBCECN_61	0	0	21	-12	15	3	26
SBCECN_70	-3	0	0	-22	5	-4	-24
SBCECN_80	0	0	0	-2	-1	0	-4
Total Rhône River Catchment	-33	43	-34	-235	-145	15	-389

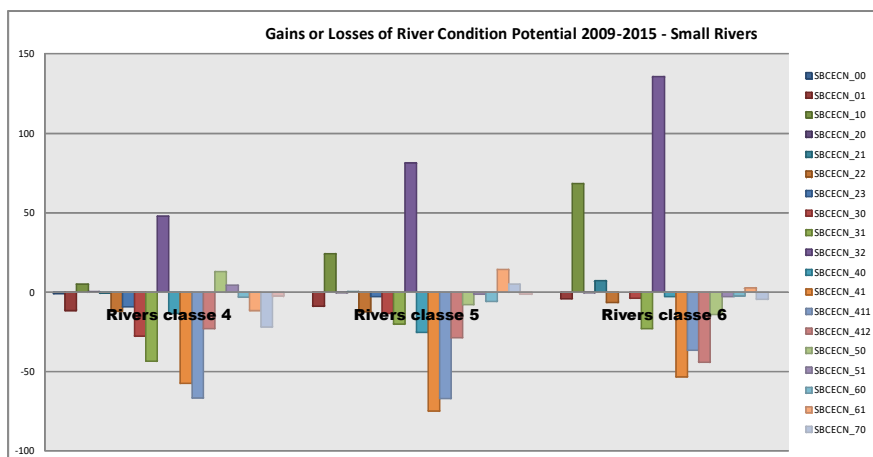


Figure 4.22. Gains or Losses of SB_ENCA Rivers Condition Potential 2009-2015 of Small Rivers (kmcn_R weighted by status index).

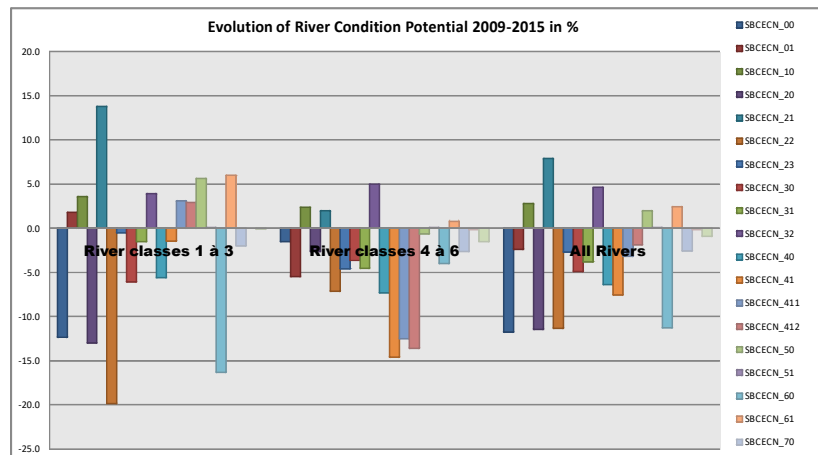


Figure 4.23. Evolution of SB_ENCA River Condition Potential 2009-2015 in %.

As shown all along the making of the rivers accounts, changes are fast and contrasting information is frequently revealed by the representations at UZHYD and ENCAT or SB_ENCA scales.

In summary, the presented results represent a short but important extract of the full water accounts and ecological condition of rivers in the Rhône watershed. They constitute the frame on which I will address in the following Chapter 5 the next step in the ENCA protocol: from River Condition Potential to Net River Ecosystem Potential.

4.6 Issues, a synthesis of the data frame

Producing the accounts implied using as much as possible existing data publicly available and assessing their fitness for the purpose of ecosystem accounting. *Ad hoc* data collection as well as modelling of raw data from monitoring stations was beyond the remits of the accounting exercise.

Considering the water accounts, data for accounting proved to be fairly abundant and of good quality, essentially accessible through the French Government portal “EauFrance” and from the website of the *Rhône-Méditerranée-Corse* Water Agency. The wide realm of water issues is well covered, including with high quality geographical data. Among the limitations we were faced with I should mention that not all data collected and hold by one or the other EauFrance’s partner organisations are accessible in digital format, fit for accounting. Some data are displayed only as printed documents or maps. Others are presently restricted for commercial or confidentiality

Chapter 4 Ecosystem Water Account

reasons, which may seem questionable in the context of open access to public data. In the event of a revision of water accounts, solutions will have to be found in such cases.

Data gaps were identified concerning water use. Regarding water abstraction, the database of fees is detailed but inconsistent over time because of changes in practice and legislation. The same difficulty is faced with wastewater treatment plants data. These issues are well known and warnings provided to users. However, they remain the only source of information on water use and discharge. For the purpose of the ENCA accounting test, corrections of obvious errors and adjustments of probable bias have been done. Better revision of these datasets could be carried out by experts of the domain, an endeavour which could be justified by the growing importance of water management issues and the interest of knowing past trends.

Water management and distribution is another area where data are not easily accessible. They exist but are held by companies or semi-public organisations which provide data on their water withdrawals (for hydroelectricity, cooling and canal water supply) but not the full picture of water transfers for distribution (in particular of drink water and irrigation water), as well as storage/destorage of water. Again, the issue is certainly not that of the existence of such data, but their accessibility.

Conclusion

In this Chapter the evaluation of the water resource focused on the intensity of use and the calculation of an index of ecosystem health. For the calculation of the index of intensity of use, we took a different approach as compared with the ENCA-QSP methodology (accessible resource / total use) due to the quality of available data. Instead, we calculated an index of the rivers condition potential ($\text{length} \times \ln(\text{runoff}/(\text{m}^3/\text{s}) \times \text{ecological status})$). An ecosystem health index was calculated based on the biochemical quality of lakes and reservoirs, rivers and water courses, snow and glaciers, soil and vegetation, and ground water availability. These indexes will serve to the calculation of water unit value and its integration to the Ecosystem Capability Unit in Chapter 6.

Supplementary file 4.1

Description of the computation modules

The ENCAT_ARBO prototype and revised ENCAT_ARBO module

Accounts computation by UZHYD

-Computation of rivers srkm values for compiling rivers quality account (reported below)

Calculation of Surface Runoff Captured by Rivers

Calculation of river runoff by UZHYD (simplified method)

Implementation of the flow calculation method to Rivers quality/ status account

Specific methodologies used for various variables

Precipitations and evapotranspiration

-WorldClim and CGIAR datasets

-SbwEWA datasets and processing

Glaciers melting and estimating stocks

Groundwater renewable resource

-Deep percolation to groundwater; swbEWA data and the issue of permeability

-Groundwater draining to rivers

-Groundwater renewable resource

-Groundwater quality

The lake and reservoirs issue

Soil water

Water use

-withdrawals of water

-Returns of water: losses, irrigation, non-consumptive uses, wastewater discharge

-The "grey water" in ENCA

Water balances and indices of water bodies' status

List of the data needed

- Stocks of water bodies (called "assets" in the SEEA, term taken up by ENCA), including soil moisture

- Precipitations

- Actual evapotranspiration

- Percolation to groundwater

Chapter 4 Ecosystem Water Account

- Emptying of groundwater to rivers
- Variation of soil moisture (important for monthly or quarterly accounts)
- Storage / retrieval of snow and ice
- Storage / retrieval of reservoirs (important for monthly or quarterly accounts)
- Artificial water transfers from one basin to another
- The (underground) flow of underground aquifers
- Direct water withdrawals by sector of use (including distribution), known by municipalities
- Water transport by distributors and supply to areas of use
- Returns of water: losses and leaks in transport, irrigation, discharges of treated and untreated waste water
- The ecological status of water bodies (component indices: chemical or biotic indexes, or synthetic indices as provided to the WFD)
- Fragmentation of rivers (obstacles to flow) if not included in the ecological status index.

The analytical units and their connection

UZHYD = the French hydrological units

ENCAT, smaller than Sub_basins as they are used to define socio-ecological units, but compatible. ENCATs have a hierarchical relation which allows cumulative assessment of runoff for their main drains

Rivers and lakes: from the national geo data (Carthage, SANDRE).

Connecting quantitative and qualitative accounts: the SRKM

Presentation/definition of the SRKM. Formula. Need to express rivers quality (weighting factor allowing comparisons of rivers of various sizes).

Chapter 5 Ecosystem Infrastructure Account

Outline

This chapter deals with the ecosystem infrastructure accounts, the last of the three core accounts in ENCA. There is no single formula to characterize and report on this ENCA component (the indicator). To evaluate the ecosystems infrastructure integrity, intermediate indexes NLEP and NREP (Net Landscape Ecosystem Potential and Net River Ecosystem Potential) are calculated for various land cover types and river ecosystems, and ultimately the indexes are combined to obtain the Total Ecosystem Infrastructure Potential. The Intensity of use is calculated as the change of the Total Ecosystem Infrastructure Potential between two successive periods. An index to diagnose Ecosystem health is also calculated by measuring changes in the state of Biodiversity.

5.1 Introduction

In ENCA-QSP, the biosphere integrity account is called “Ecosystem Infrastructure Functional Services Account” to highlight that services cannot be measured directly as in bio-carbon and water accounts, but indirectly as a potential access to ecosystems functions. Biosphere integrity accounts are composed of two sets of accounts: quantitative accounts and qualitative or semi-quantitative estimations of ecosystem health.

Quantitative accounts build upon spatial data representing ecosystem accounting units and typical features like type and properties. Semi-quantitative estimations are carried out in the Table of Intensity of Use and Health indexes for taking into account the ecosystem health aspects that are not captured in the quantitative tables, in particular the change in species diversity.

Biotic parameters and their relationships within ecosystems, is a domain with a reputation of complexity and lack of data. Further to their intrinsic value and to any direct economic value they might have, species and habitat types are linked to ecosystem functions and /or processes, which enable the provisioning of ecosystem services. From photosynthesis and primary production to individual species and populations and further on to the formation of vegetation and habitat types, biotic parameters are a dynamic component of ecosystems. For ecosystem capital accounting, changes in all these levels are important as indications of health and resilience.

Chapter 5 Ecosystem Infrastructure Accounts

Ecosystem capital accounts are focusing on identifying degradation, and as such they do not require complete balanced accounts for stocks and flows of all species and habitat types. What matters is to take stock of change in a meaningful way, either in numeric values, based for example, on the loss of habitat for a set of species, or in generic assessments such as good, unfavourable or bad conservation status of a set of species.

Further to the conceptual framing of biodiversity and ecosystem services, nature conservation and biodiversity policies need to ensure that degradation is properly addressed and progress towards targets can be measured. It is also essential to be able to re-use data and information from EU reporting into ecosystem management indicators.

This work follows the main guidelines of Ecosystem accounts as described in the Ecosystem Infrastructure Functional Services of the CBD TS77 Technical Series, Chapter 7 (**Weber, 2014**), as shown in **Figure 5.1**.

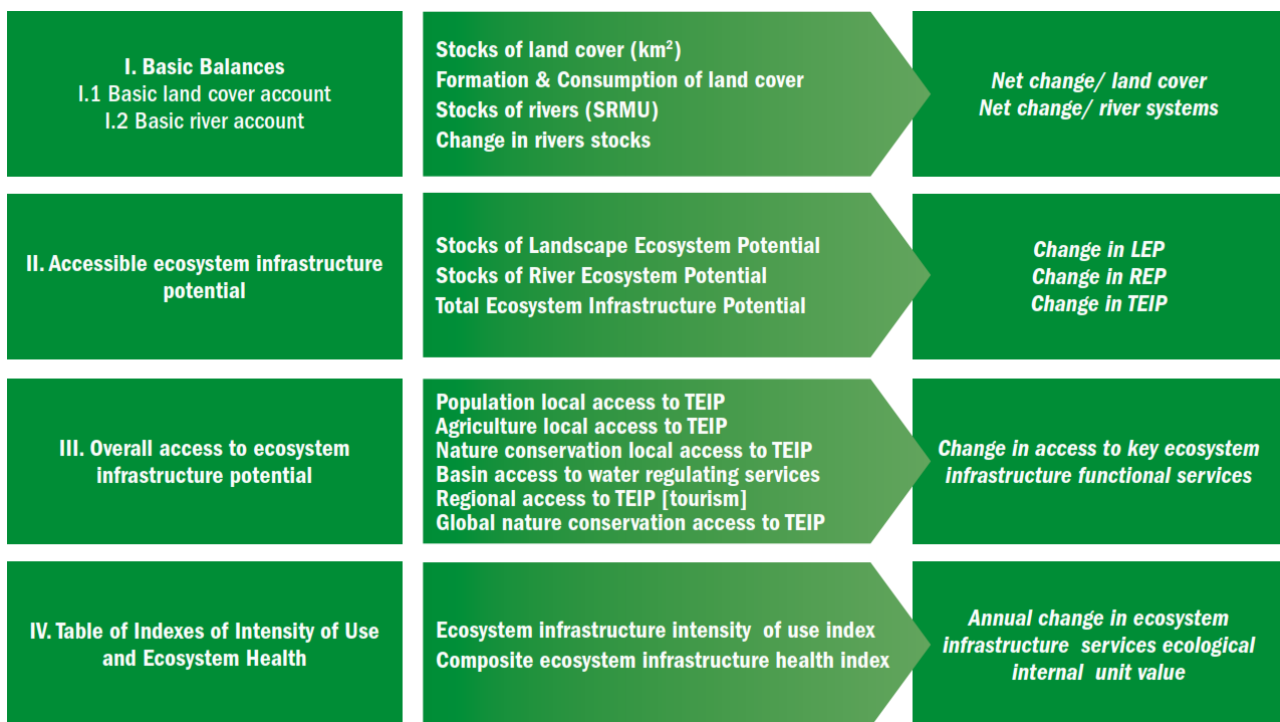


Figure 5.1. Structure of ENCA-QSP Ecosystem Infrastructure Functional Services Account.

More specifically, the Ecosystem Infrastructure Account is articulated in the ENCA protocol as shown in **Figure 5.2**.

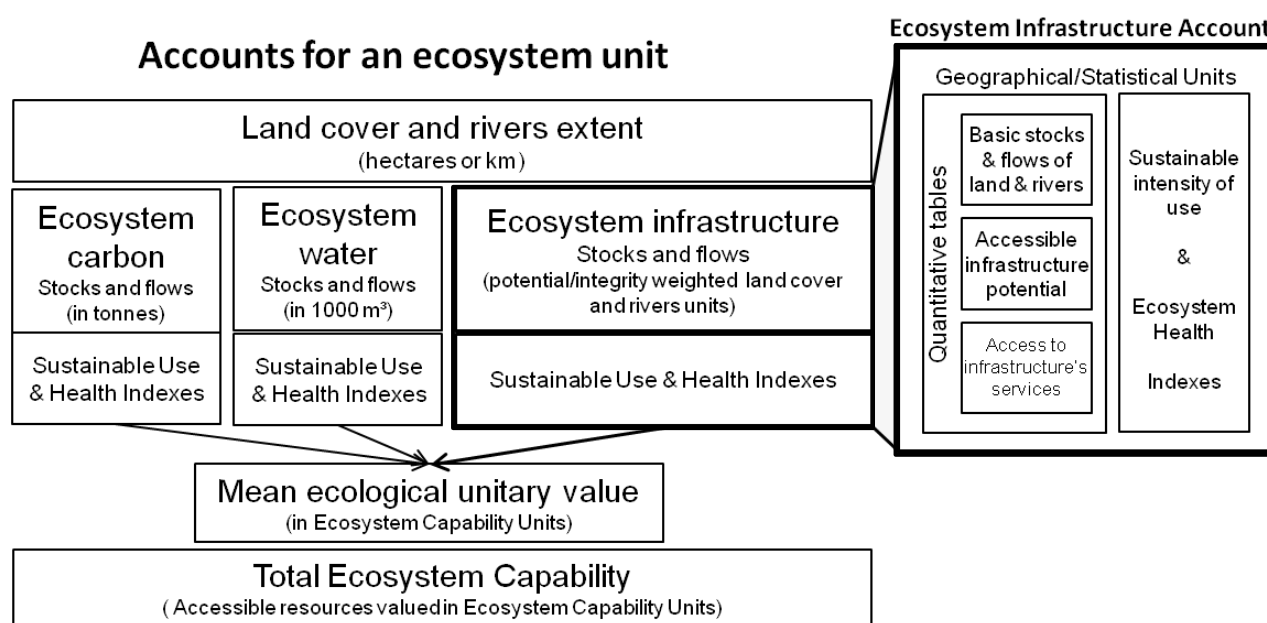


Figure 5.2. Articulation and integration of ENCA components. This Chapter addresses Ecosystem Infrastructure Account and related synthetic indicators. As opposed to ecosystem bio-carbon and water accounts which directly record quantities measured (or estimated) in resp. tonnes and thousands of m³, the ecosystem infrastructure account relates to intangible services which can only be quantified in an indirect way. This indirect quantification is based on the accessible potential and integrity of the ecosystem infrastructure, landscapes, and rivers. As rivers are measured as linear elements weighted by their mean water discharge, Net Rivers Ecosystem Potential is firstly assessed separately from the Net Landscape Ecosystem Potential and then merged with it as value per hectare. The Use table is also specific as only a few (important) services are proposed in the ENCA manual, with no aggregation. In principle, indexes of change in species biodiversity are recorded in the table of Ecosystem health. However, because the synthetic indexes on rivers ecological status provided to the Water Framework Directive include biodiversity, there is a slight difference in presentation regarding landscape and rivers..

The account of ecosystem infrastructure functional services is composed of three elements: landscape systems, rivers systems, and marine coastal systems. The latter have not been addressed in this project.

Landscape systems are linked to land cover accounts (see **Chapter 2**). They aim at producing an assessment of *Net Landscape Ecosystem Potential* (NLEP). NLEP combines an index of greenness, an index of landscape fragmentation, and an index needed to capture the high nature value of particular ecosystems (Weber et al, 2008). This high nature value is assessed regarding the concrete attention given to habitats, fauna, and flora by scientists and by the environmental agencies which translate their knowledge into action by means of various protection measures. These nature protection zones are one expression of the value that the society gives to biodiversity. This means that if it happens that a protected area is degraded, the estimated loss is higher than that of common

Chapter 5 Ecosystem Infrastructure Accounts

landscape. From the documentation available, and in particular the maps of all protection types, an index has been constructed to measure high nature value. This index is called NATURILIS, “LIS” being the reference to “lissage”, the French word for smoothing. Indeed, the idea is that the influence of one protected particular area goes beyond the strict perimeter, as it contributes to the neighbouring landscape by providing niches and stepping stones for corridors. Symmetrically, a protected area may be under stress from its neighbourhood, cities, intensive agriculture or forestry, etc. This is particularly true for small areas which are vulnerable. All that has justified the use of smoothing techniques for the construction of NATURILIS.

For river systems, the *Net River Ecosystem Potential* (NREP), includes the River Condition Potential index (length, water discharge, and ecological status parameters), and rivers high nature value derived from nature protection (NATRIV) (see **Chapter 4**).

Of note, ENCA synthetic indicators have been conceived as means to *characterize the state of integrity of various land cover types and river ecosystems*. In **Box 5.1** I attempted integrating the main notions that participate in the conception of these synthetic indicators.

Box 5.1. Integrating main notions of integrity, potential, capability, no net ecosystem degradation, ecosystem capital debt, and the issue of ecological value, economic value and "price".

NLEP and NREP characterize as a matter of fact a potential, or capability, of a given ecosystem / landscape category to maintain its functions and provide the range of services that they have the potential to supply, e.g. the indicators serve to describe the evolution of such a potential. In that sense, the potential approach has some similarities with the HANPP concept (Haberl et al, 2007) where potential relates to the amount of the biomass flow prone at being appropriated by human activities and by nature for its renewal. HANPP is computed from land cover data both considering NPP modelling and the assessment of appropriation, in particular by land use and agriculture and forestry harvests. Most variables needed for HANPP calculations are available in ENCA ecosystem carbon account. However, an important difference between the two approaches is that HANPP relates to a theoretical potential corresponding to a climax situation while ENCA target is of accrual type and refers to no net degradation during an accounting period, and proposes to start from a recent date (e.g. 1990, the baseline year of the United Nations Framework Convention on Climate Change for industrialized countries to returning their greenhouse-gas emissions by the year 2000).

Potential is deliberately associated in ENCA with

- the response capacity of ecosystems estimated in terms of health status that measures the evolution of their resilience;
- the landscape dimension and the intrinsic injunction of no net degradation of ecosystems.

The resulting frame allows translating the potential indexes into Ecosystem Capital debts and corresponding costs (cf. Chapter 1, also see Chapter 6). In that respect,

- integrating land and river ecosystems constitutes a strong signal: river systems can be considered as a particular land cover type measured in terms of length instead of area;
- evaluating the ecological debt that is bound to the objective of preserving the net ecological value of ecosystems (e.g. no net degradation) by taking into account the full cost of ecosystem maintenance or re-vegetation.

In this context, in the ENCA terminology and practice the notion of "price" is employed metaphorically to address the issue of Internal Unit Value of given resource categories (see Chapter 6). The latter is the ecological value equivalent to the notion of price for the economic value. The ecological value, as opposed to the economic value, targets the long-term functions of the ecosystem as measured through their potential or capability. This allows to envisage the amortization of the ecosystem capital degradation through accounting norms / standards that can register the ecological debt in a synthetic unit, as I will show in Chapter 6.

5.2 Methods

5.2.1 Basic Balances Parameters and Accessible Ecosystem Potential Indexes

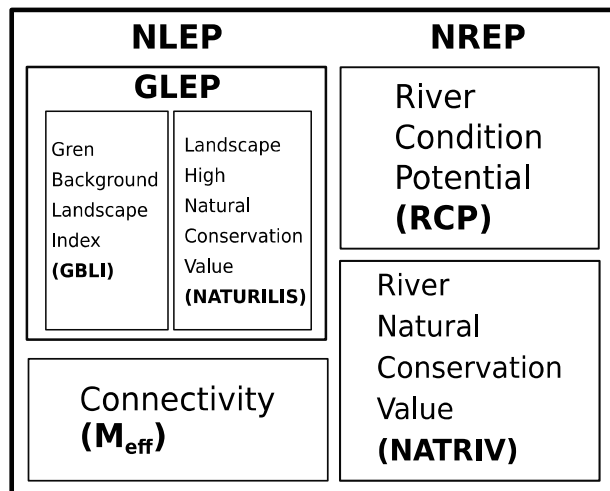
Primary inputs for evaluating the Ecosystem Infrastructure Potential are land cover (see **Chapter 2**) and rivers (see **Chapter 4**). The Total Ecosystem Infrastructure Potential is calculated here through two composite indexes, Landscape Ecosystem Potential and the River Ecosystem Potential.

The Net Landscape Ecosystem Potential (NLEP), Net Rivers Ecosystem Potential (NREP) and Net Marine Ecosystem Potential (NMEP, not considered here) are composite indicators.

Box 5.1 details the insert from Figure 5.2 and is a guideline for the reader by showing how the chosen indicators are articulated and integrated. Taken together, they constitute the landmark of ENCA.

Box 5.2. Land and River system indicators: comparative outlook and integration.

Total Ecosystem Infrastructure Potential



Net Landscape Ecosystem Potential NLEP

The NLEP, shown in **Figure 5.3**, is a macro indicator derived from land accounts and landscape characteristics. It integrates the land cover artificiality/naturalness, the areas representing high species/habitats diversity, and the landscape connectivity to measure and assess ecosystem integrity (Weber et al, 2008). The formula for the year 2000 is:

$$NLEP = GLEP \times \ln meff$$

m_{eff} , the ‘effective mesh size’, is the expected size of the area that is free of physical barriers inside the reporting units. Thus, more barriers in the landscape lower the probability that two points will be connected and lower m_{eff} (Moser et al., 2007).

To calculate the *NLEP 06* and *NLEP 12*, I use the formulas:

$$NLEP\ 2006 = \frac{NLEP2000}{GBLI2000} \times GBLI2006$$

$$NLEP\ 2012 = \frac{NLEP2000}{GBLI2000} \times GBLI2012$$

The result was divided by 100 to have values between 0.1-1 for its further integration with the Net River Ecosystem Potential.

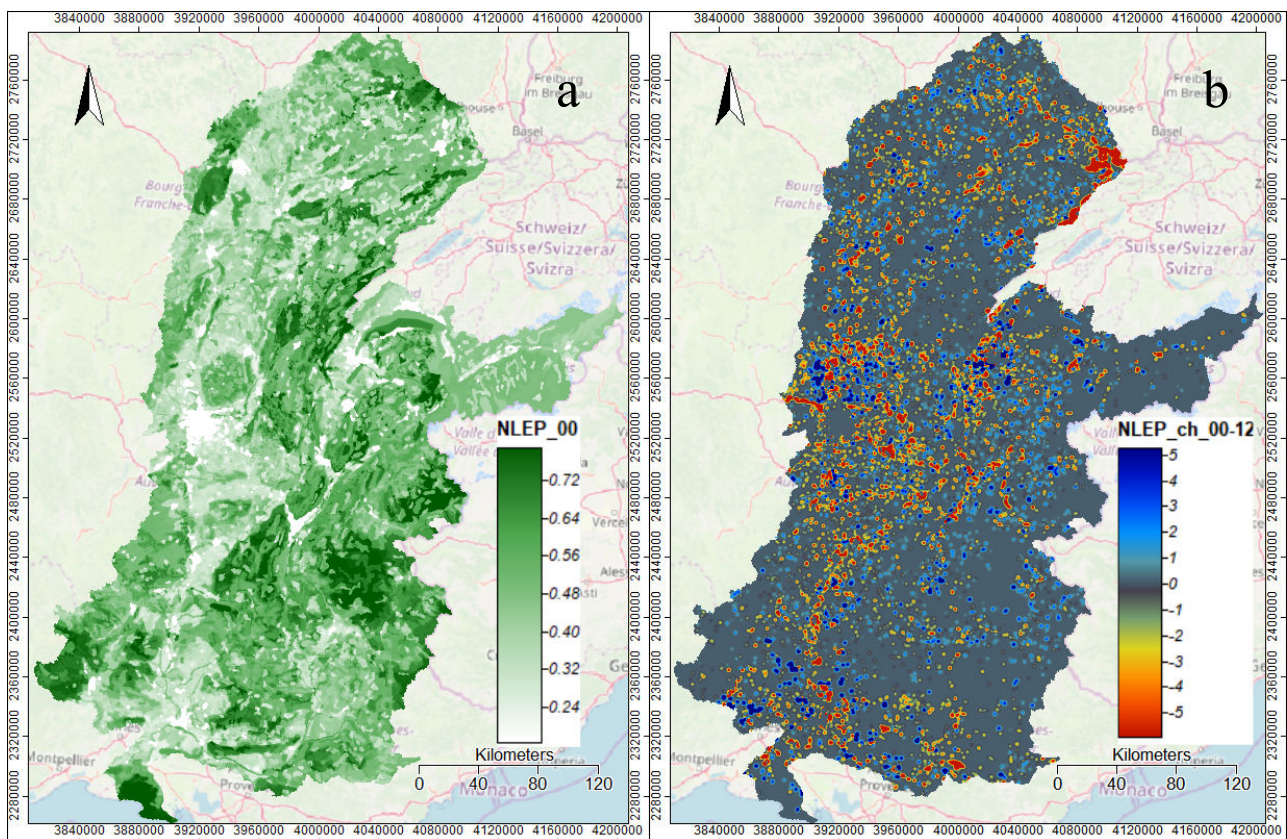


Figure 5.3. Net Landscape Ecosystem Potential for 2000 (a) is the integration of the Green Background Landscape Index (Figure 5.4), NATURILIS (Figure 5.5), and the Landscape connectivity index (Figure 5.6) with a range between 0 (white) and 1 (darkest color). **NLEP changes (%) from 2000 to 2012 (b)** with a range of -5 (darkest red) to 5 (darkest blue) . QGIS and SAGA-GIS geo-processing tools were used.

The Gross Landscape Ecological Potential GLEP is the product of two parameters, namely the land cover quality (artificiality/naturalness) and conservation areas with ecological value.

The integration of the two indexes, Green Background Landscape Index (GBLI) and the Landscape high nature conservation value index NATURILIS was done using SAGA-GIS *Grid Calculator* with the formula:

$$GLEP = GBLI \times NATURILIS$$

Green Background Landscape Index GBLI

The green background is an index map that shows “ecological potential”. It is based on the spatial distribution of natural and seminatural land (Haines-Young et al 2006). This is calculated for 2000, 2006, and 2012.

The Dominant Land Cover Types (DLCT) file (*see Carbon chapter, section 3.2.4*) was used as input to calculate this index. Each type was separated and re-coded to a binary outcome to calculate the sum of the multiplication of the 10 DLCT by a weight from 0 to 1 according to their intensity of land use as deduced from land cover (**Figure 5.4**). The operation was done in SAGA-GIS *Grid calculator* using the scores:

artificial areas = 10
agricultural land = 20
pastures = 60
agricultural mosaics = 60
forest = 100
natural grassland = 80
shrubland and heathland = 80
open spaces = 80
glaciers and perpetual snow = 80
wetlands and water bodies = 80

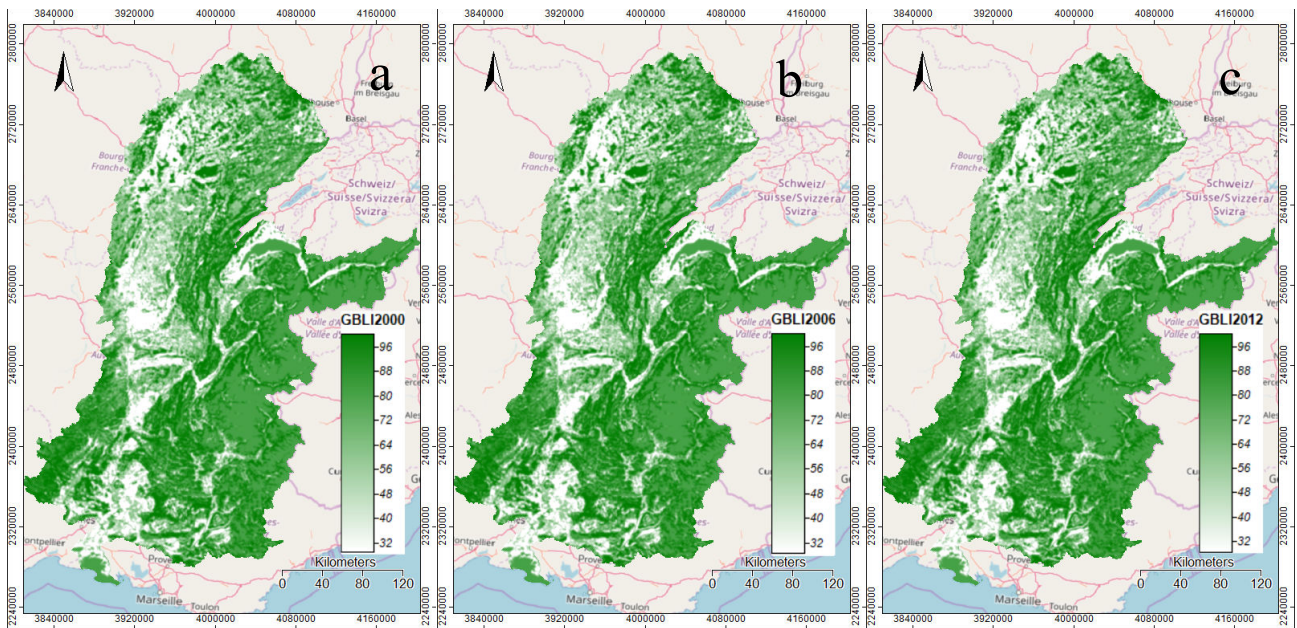


Figure 5.4. Green Background Landscape Index 2000 (a), 2006 (b), 2012 (c) calculated from the DLCTs (Figure 3.13) scored according to their artificiality/naturalness with a range from <30 (white, most artificial) to 100 (darkest color, most natural).

The GBLI scores are conventional. Sensitivity tests have been done considering GBLI change. Finally, the main issue with GBLI relates to forests which are not discriminated in CORINE land cover according to their naturalness. Therefore, plantations of poplars will be given the same rating as old oaks forests. This bias is partly corrected by the NATURILIS index. However no distinction will be done out of protected areas between more or less intensively managed forests.

Landscape high nature conservation value index NATURILIS

The nature conservation value is based on the interpretation of protected areas designations adopted by governments through scientific recommendations. Different years were combined.

The international and French designations were downloaded from the *Inventaire National du Patrimoine Naturel* (INPN, 2018), with 16 files of designated areas and buffers covering the study area (**Table 5.1**).

Table 5.1. Protected areas and their corresponding file codes (in French)

	French name	code	year
1	Arrêtés de protection de biotope APB	apb2018_03 / N ENP APB S 000	2018
2	Parcs nationaux PN	pn / N ENP PN S 000.shp	2016
3	Réserves biologiques RB	rb/ N ENP RB S 000.shp	2016
4	Réserves de biosphère MAB	bios2018_01 / N ENP MAB S 000.shp	2018
5	Réserves intégrales de parc national RIPN	ripn/ ripn2013.shp	2013

Chapter 5 Ecosystem Infrastructure Accounts

6	Réserves naturelles nationales RNN	rn2018_03/ N ENP RNN S 000.shp	2018
7	Réserves nationales de chasse et faune sauvage RNCFS	rncfs/ N_ENP_RNCFS_S_000.shp	2015
8	Réserves naturelles régionales RNR	rn2018_03/ N ENP RNR S 000.shp	2018
9	Sites acquis des Conservatoires d'espaces naturels CEN	cen/ cen2013_09.shp	2013
10	Sites du Conservatoire du Littoral SCL	cdl2018_03/ N ENP SCL S 000.shp	2018
11	Zone humide protégée par la convention de Ramsar RAMSAR	ramsar_2018_01/ N ENP RAMSAR S 000.shp	2018

Natural Areas of Ecological Interest of Fauna and Flora (ZNIEFF, in French)

12	ZNIEFF1 Continentales	znieff/ znieff 1	2018
13	ZNIEFF2 Continentales	znieff/ znieff 2	2018

Areas of importance for bird conservation (ZICO, in French)

14	Zone d'importance pour la conservation des oiseaux (ZICO)	zico/ zico.shp	1994
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NATURA 2000

15	Sites classés au titre de la Directive Habitats : périmètres transmis à la CE (ZSC/pSIC/SIC)	sic/ sic.shp	2018
16	Zones de Protection Spéciale (ZPS)	zps_2017/ zps1712.shp	2017

The files were reprojected to the ETRS89 Lambert Azimuthal Equal-Area projection in QGIS. The new field “protection” was created and populated with the protection scores according to their management objectives (*Table 5.2*), then transformed to 250 m cell size raster file using the Rhône buffer grid system in SAGA-GIS *Shapes to grid*.

Table 5.2. Protection scores. Abbreviations are in French: **APB**: Arrêtés de protection de biotope, **RB**: Réserves biologiques, **RIPN**: Réserves intégrales de parc national, **RNCFS**: Réserves nationales de chasse et faune sauvage, **RNR**: Réserves naturelles régionales, **CEN**: Sites acquis des Conservatoires d'espaces naturels, **SCL**: Sites du Conservatoire du Littoral, **RNN**: Réserves naturelles nationales, **PN**: Parcs nationaux, **ZICO**: Zone d'importance pour la conservation des oiseaux, **SIC**: Sites classés au titre de la Directive Habitats **ZPS**: Zones de Protection Spéciale, **RAMSAR**: RAMSAR sites. **ZNIEFF1**: Zones Naturelles d'Intérêt Ecologique Faunistique et Floristique de type 1 (secteurs de grand intérêt biologique ou écologique), **ZNIEFF2**: Zones Naturelles d'Intérêt Ecologique Faunistique et Floristique de type 2 (grands ensembles naturels riches et peu modifiés, offrant des potentialités biologiques importantes).

Protection level	Name code	Protection score
Strong protection	APB, RB, RIPN, RNCFS, RNR, CEN, SCL, RNN	10
National parks	PN	6
European status	ZICO, SIC et ZPS	3
RAMSAR	RAMSAR	3
Natural area of ecological interest	ZNIEFF1 et ZNIEFF2	2
Others	Zones de gestion (MAB) et autres territoires	1

The 16 designations of protection were added up in SAGA-GIS *Grid calculator*. The result was smoothed with the parameters 3 and 10 for standard deviation and radius, respectively, to get the potential in the neighborhood, and resampled to 1-ha pixel-size with SAGA-GIS *Resampling*, B-Spline interpolation method (**Figure 5.5**). The normalization resulted in a scale from 0 to 5.

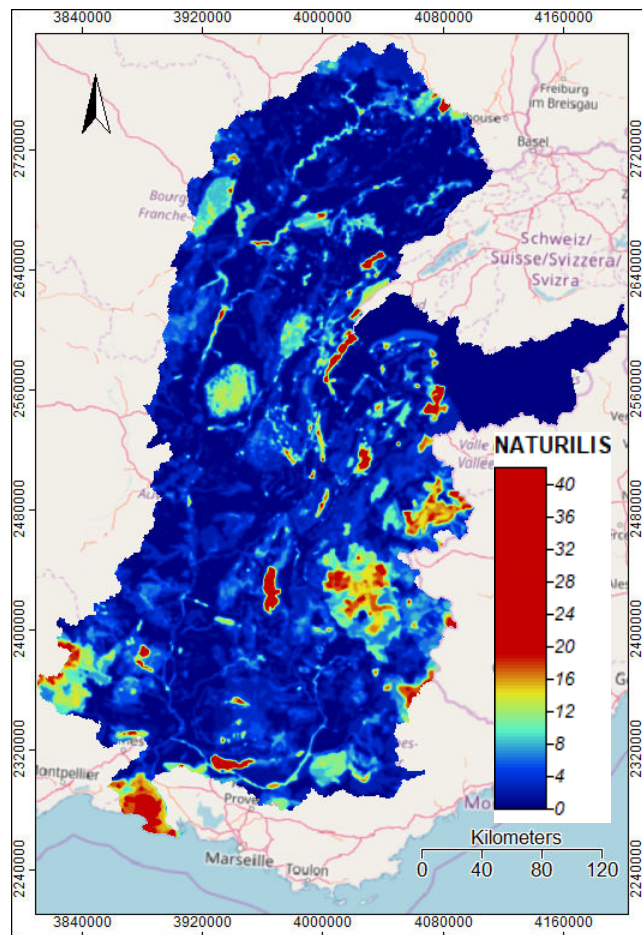


Figure 5.5. The Natural Conservation Value (NATURILIS) computed as the sum of datasets on protected areas from the Inventaire National du Patrimoine Naturel assigning them a score according to their protection status. Values >6 (from green to red) mean that the area has at least three weak designations of protection or is a national park.

Landscape connectivity index (m_{eff})

This index is calculated using the effective mesh size (m_{eff}). The m_{eff} is the expected size of the area that is free of physical barriers inside the reporting units. Thus, more barriers in the landscape lower the probability that two points will be connected and lower m_{eff} (Moser et al., 2007). The formula is:

Chapter 5 Ecosystem Infrastructure Accounts

$$m_{\text{meff}}^{\text{CBC}} = \frac{1}{A_{\text{total}}} \sum_{i=1}^n A_i \cdot A_i^{\text{cpl}}$$

where A_{total} = the total area of the reporting unit, n = the number of patches, A_i = size of patch i inside the boundaries of the reporting unit ($i = 1, 2, 3, \dots, n$), A_i^{cpl} = the area of the complete patch that A_i is a part of, i.e., including the area on the other side of the boundaries of the reporting unit up to the physical barriers of the patch (Moser et al., 2007).

Were included in the calculation, the railway, the motorway and the primary highway networks from the OpenStreetMap (OSM), and the Socio-Ecological Landscape Units (SELU, see Chapter 5, section) of the year 2000. The calculations were conducted as follows:

- 1) OSM files were generated by the modification of places and traffic ways, cf. **Appendix F**¹.
- 2) The networks were merged and reprojected to the ETRS89/ETRS-LAEA Coordinate Reference System in QGIS using *merge shapefiles* and *reproject layer*, respectively.
- 3) The result was merged with the Rhône watershed line perimeter using the same tool,
- 4) A 10 meter buffer was created in QGIS,
- 5) The SELU vector file was split with the buffer using SAGA-GIS *Symmetrical difference*,
- 6) The area of the remained fragments was calculated with QGIS *field calculator*,
- 7) The neighbors of the fragments were searched, and their areas were added up with a python script in QGIS (**Appendix G**¹).
- 8) To obtain $A_i \times A_i^{\text{cpl}}$, the areas of the fragments and those of the neighbors in each unit were multiplied using a function in QGIS *field calculator* (**Appendix H**¹).
- 9) The values of the fragments of each SELU were added up using *Group Stats* in QGIS to obtain

$$\sum_{i=1}^n A_i \cdot A_i^{\text{cpl}}. \text{ The result was saved as a .csv format, joined with its corresponding original SELU}$$

and divided by the area of the SELU.

- 10) The result was transformed to a raster format in SAGA-GIS *Shapes to grid* using the Rhône grid system 1-ha pixel-size and 250 m pixel-size (**Figure 5.6**).

- 11) The ln was calculated using SAGA-GIS *Grid calculator* for its further integration with the Gross Landscape Ecological Potential.

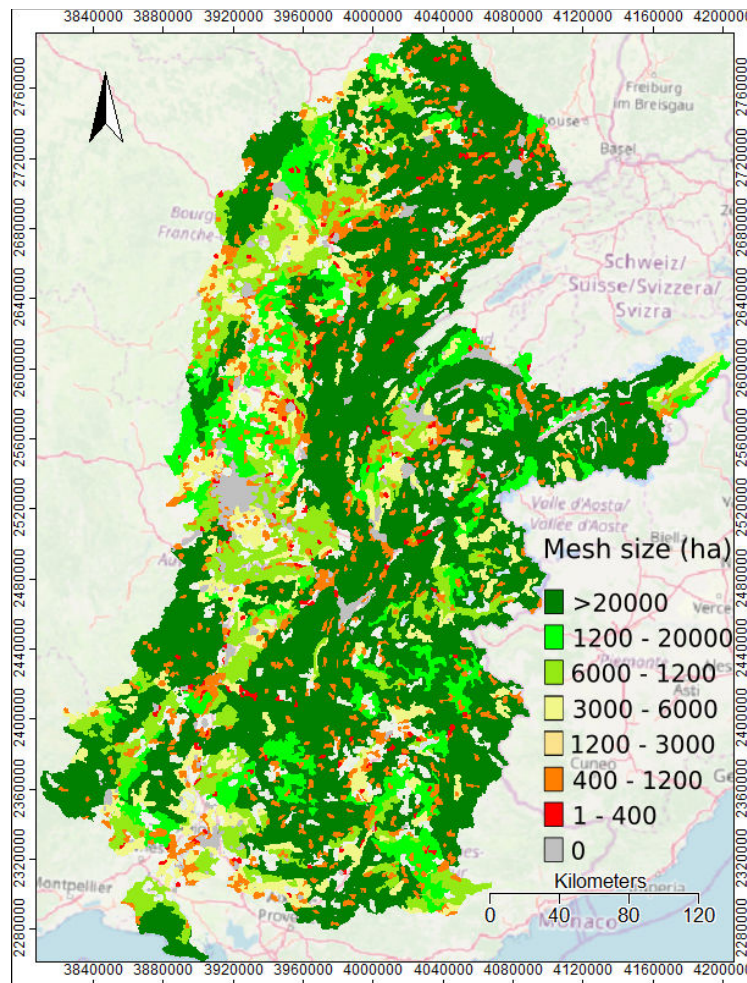


Figure 5.6. Effective mesh size m_{eff} by SELU. The m_{eff} is based on the probability that two points chosen randomly in a region will be connected (Mosser et al., 2007). SELUs are clipped with fragmentation elements (motorways and railways) from Open Street Map and m_{eff} is calculated using Python, QGIS and SAGA-GIS geoprocessing tools. The grey color indicates lowest m_{eff} (0 ha) and green the highest (> 20,000 ha).

Net River Ecosystem Potential NREP

The NREP was calculated in each pixel i as the River Condition Potential, also called River Ecosystem Background, multiplied by the NATRIV (Rivers Nature Conservation Value), the index of rivers in protected areas. To obtain values between >0 to 1, I normalized by the maximum value of this multiplication for the whole Rhône watershed area (**Figure 5.7**):

$$NREP_i = \frac{(RCP_i \times NATRIV_i)}{\max(RCP \times NATRIV)}$$

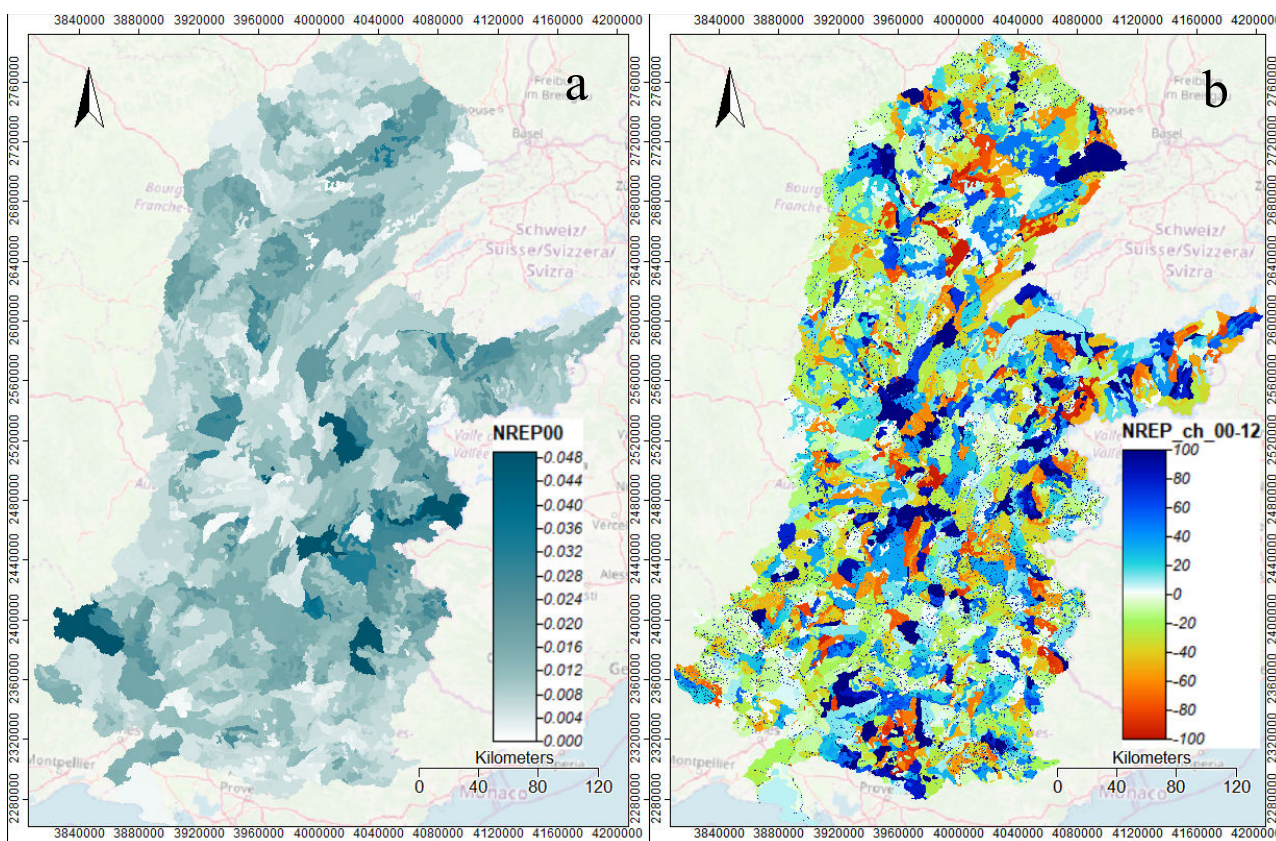


Figure 5.7. Net River Ecosystem Potential 2000 (a) is calculated as the River Condition Potential (Figure 5.8), also called River Ecosystem Background, multiplied by the NATRIV (Figure 5.9). The dark colors indicate higher potential levels. The range of the index is zero to one. The values are very small due to extreme values in a few pixels in comparison to the average (see Figure 5.8). **(b) NREP changes (%)** from -100 (darkest red) to 100 (darkest blue).

River Condition Potential RCP

Rivers Condition Potential is based on the *srkm_Rivers* unit. This unit includes length and a quantitative dimension related to water discharge. The unit is weighted by the ecological status index (EEA, 2018) of Hydrological Units (UZHYD). The ecological status assessment combines three types of parameters: chemical, biological, and morphological (including fragmentation) (see **Chapter 4, section 4.5.2**).

$$RCP = srkm_R \times status\ index$$

The “Ecological status is based on biological quality elements and supporting physico-chemical and hydromorphological quality elements.” (EEA, 2018, page 7).

The five state classes from Cartograph 'EauFrance (2015) were converted into notes according to the rule that a downgrading of a class results in a relative loss of 25%, the very good quality having a score of 1. After rounding, the following scoring grid is obtained:

Very good = 1, Good = 0.8, Average = 0.6, Poor = 0.4 and Bad = 0.3.

The values in 2000, 2006 and 2012 are interpolated from the original values in 2009 and 2015 based on annual change and taking the `srkm_R` as maximal allowed value (**Figure 5.8**).

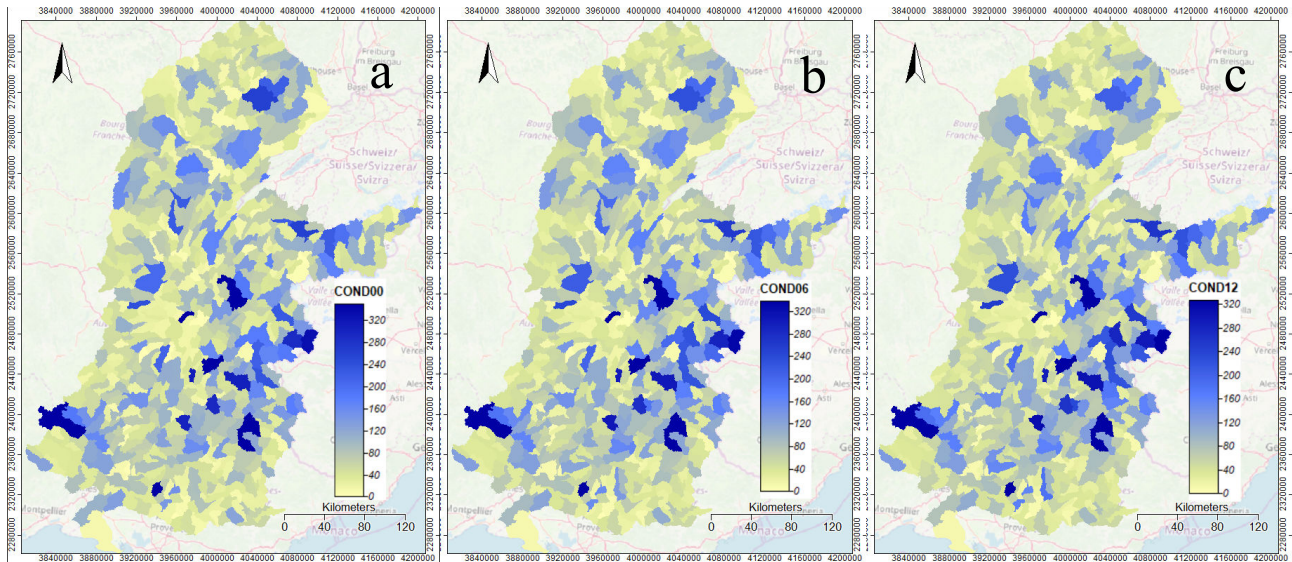


Figure 5.8. River Condition Potential 2000 (a), 2006 (b), 2012 (c). Cool colors indicate the values above the average.

River Natural Conservation Value NATRIV

NATRIV is produced with the same input data as NATURLIS but without gaussian smoothing as the intrinsic river high nature values is now targeted. Rivers (BD Carthage, see Data sources) are rasterized and the file is intersected with the NATURLIS primary data which has been normalized to rank from 1 to 5. The work has been done by class of rivers, and then the 6 classes have been summed up. The baseline file of rivers was assigned the value 1 for rivers outside designated protected areas. The product of the intersection is added to give finally a rating scale of 1 to 6. **Figure 5.9** illustrates the effect of the NATRIV procedure.

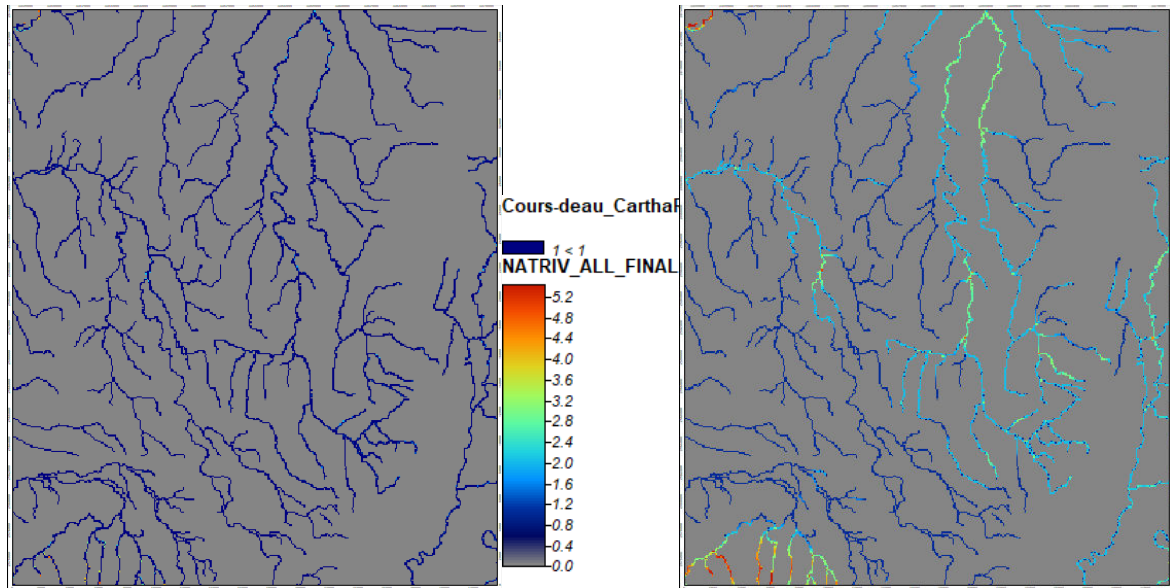


Figure 5.9. Rivers from BD Carthage (left) and Rivers scored with a normalized NATURILIS index (Figure 5.5) to obtain the NATRIV index (right). SAGA-GIS geoprocessing tools were used.

5.2.2 Total Ecosystem Infrastructure Potential (TEIP)

The NLEP and NREP indicators are aggregated by Socio-Ecological Landscape Units (SELU) to compute the Total Ecosystem Infrastructure Potential (TEIP) (Figure 5.10).

$$TEIP = \frac{\left(\sum_i^{pixelsNLEP} NLEP_i + \sum_i^{pixelsNREP} NREP_i \right)}{\left(\# pixelsNLEP + \# pixelsNREP \right)}$$

The pixel values and the number of pixels were added up by SELU polygons using the Module Grid Statistics for Polygons in SAGA-GIS.

The TEIP was calculated in QGIS field calculator with the formula:

(“NLEPt” + “NREPt”) / (“no. pixels NLEP” + “no. pixels NREP”)

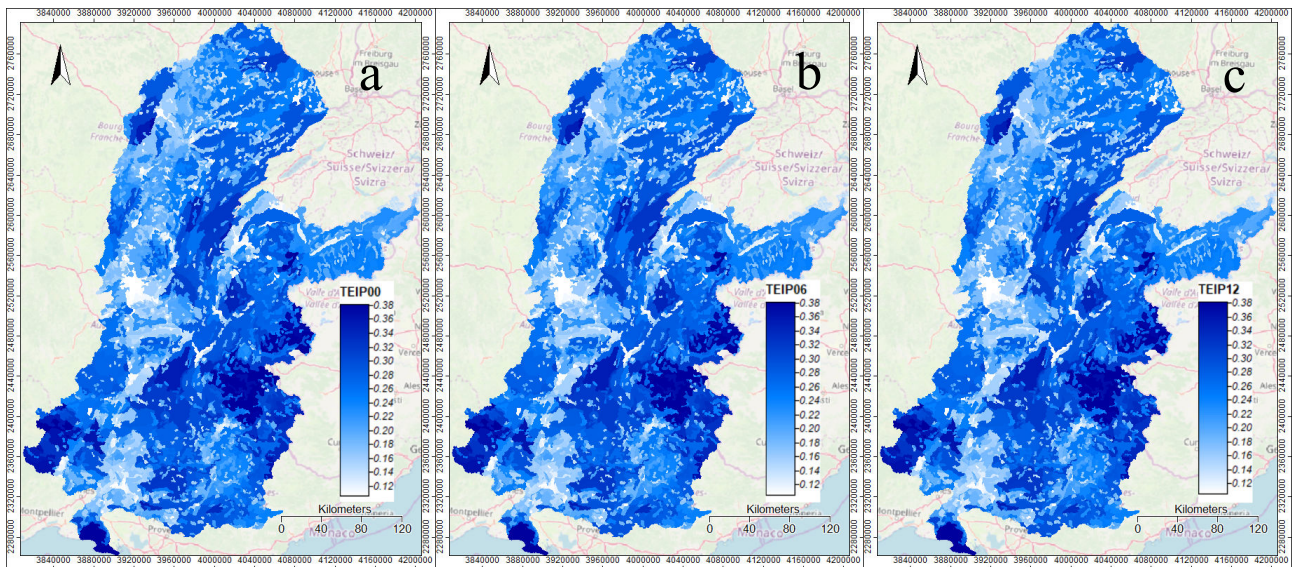


Figure 5.10. Total Ecosystem Infrastructure Potential 2000 (a), 2006 (b), and 2012 (c). This is calculated as the integration of NLEP (Figure 5.3) and NREP (Figure 5.7) indexes using SAGA-GIS geoprocessing tools. Dark colors indicate higher values. The theoretical range is from 0 to 1, where 0.5 indicates medium Total Ecosystem Infrastructure Potential.

5.2.3 Indexes of Intensity of use and Ecosystem Health

Index of Intensity of Use

This index was calculated as the change in TEIP values in 2000, 2006, and 2012 (**Figure 5.11**). The yearly change from 2000 to 2006 and from 2006 to 2012 were calculated according to the following formulas:

$$\text{Change TEIP 2000} = \frac{(\text{TEIP2006} - \text{TEIP2000})}{6}$$

$$\text{Change TEIP 2012} = \frac{(\text{TEIP2012} - \text{TEIP2006})}{6}$$

$$\text{Change TEIP 2006} = \frac{(\text{Change TEIP2000} + \text{Change TEIP 2012})}{2}$$

To maintain the range of the index between >0 and 1, the changes have been rescaled by multiplying by six, adding 1 and then dividing by two using the formulas in QGIS field calculator:

$$\text{Change TEIP 2000} = ((\text{"TEIP2006"} - \text{"TEIP2000"}) + 1) / 2$$

$$\text{Change TEIP 2012} = ((\text{"TEIP2012"} - \text{"TEIP2006"}) + 1) / 2$$

$$\text{Change TEIP 2006} = (\text{"Change TEIP 2000"} + \text{"Change TEIP 2012"}) / 2$$

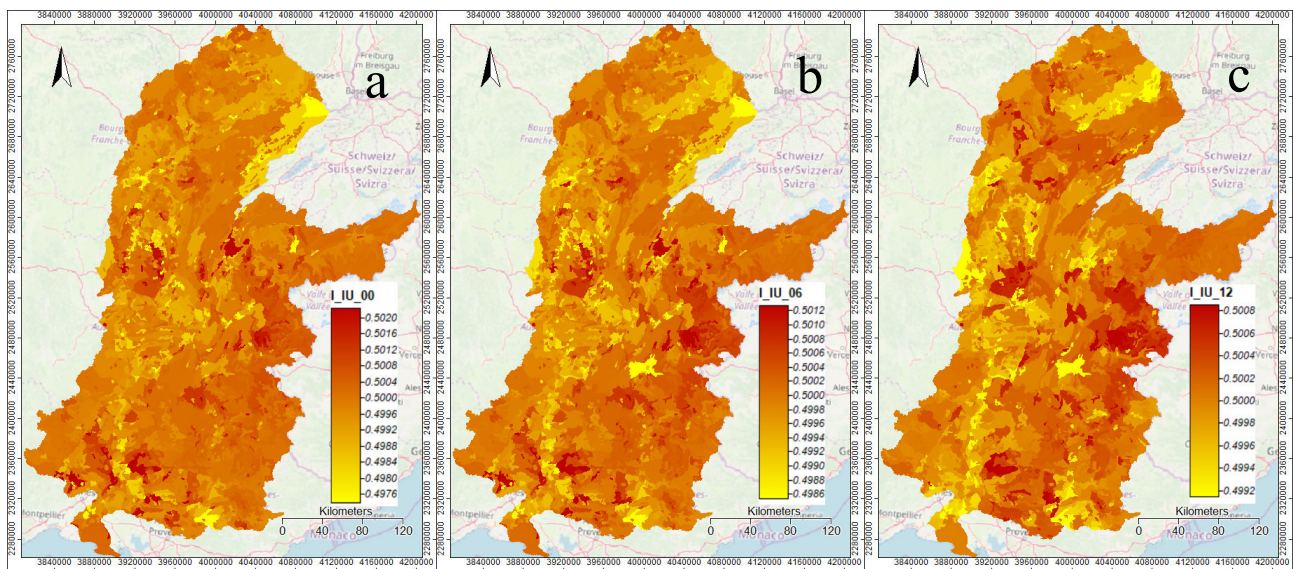


Figure 5.11. Intensity of use 2000 (a), 2006 (b) and 2012 (c). This is calculated as the change in TEIP (Figure 5.10) using SAGA-GIS geoprocessing tools. The value of 0.5 means no change, > 0.5 improvement (darker color), and < 0.5 degradation (lighter color).

Index of Ecosystem Health

A Biodiversity index was used to evaluate the Ecosystem Health. It was calculated using the methodology tested at the European Environment Agency in 2012. This is based on Article 17 reporting to the NATURA2000 EU Habitats Directive. Monitoring of conservation status is an obligation arising from Article 11 of the Habitats Directive for all habitats (as listed in Annex I) and species (as listed in Annex II, IV and V) of Community interest. Consequently, this provision is not restricted to Natura 2000 sites and data need to be collected both in and outside the Natura 2000 network to achieve a full appreciation of conservation status. The main results of this monitoring have to be reported to the Commission every six years according to Article 17 of the directive. Data is available in BISE, the Biodiversity Information System of Europe run by the European Commission and the EEA with the support of its European Topic Center on Biological Diversity (ETC-BD). The 2006 set of data processed by the EEA has been used. No update has been done with 2012 Article 17 data.

Article 17 requires Member States to report for designated habitats and for circa 1200 species of Community interest. Only species data have been used for the EEA biodiversity test accounts. Species reporting is organised according to a 10 km x 10 km grid. To each individual species record is attached an expert judgement on 4 parameters: Population, Suitable habitat (area of distribution), Range (occupation of the area of distribution) and Future prospects. For each of these parameters, the expert judgement is either “Favourable”, “Unfavourable-Inadequate, Poor”, “Unfavourable-

Bad” or “Unknown”. It can be anticipated that the surveyors would provide more robust information for “Population” and “Future prospects”. Therefore, the test uses only these two indexes. The methodology consisted in matching species (from the report in 2006) to Dominant Land Cover Types. For each 1 km² cell of the Dominant Land Cover Types, two indexes are calculated for Population (POP) and Future trends (FUT). Their format is Favourable (increasing or stable), Unfavourable (bad or poor) or Unknown (with a value of zero).

For Europe, 1 km² pixel values range from -85 to +66, according to ecosystem types, biogeographical regions, and countries. These values correspond to samples of populations of species. When samples are of very uneven sizes, they need to be normalised for the sake of comparison, in particular regarding the trends that they measure. An established way of doing it is to convert sample values into their logarithm in order to measure proportional changes. The calculation procedures below are based on Weber and Spyropoulou 2010, the ENCA-QSP, and JL Weber communications.

For 2006, the value of each pixel is $\ln(\text{POP}) + \ln(\text{FUT})$. In fact, as \ln is undefined for 0 and for negative values, the calculation has to be done in several steps:

- a) Extraction of the positive values and calculation of corresponding $\ln+1$ values;
- b) Extraction of the negative values, multiplication by -1 and calculation of corresponding $\ln+1$ values;
- c) Subtraction of b) values to a) values to obtain the 2006 Species biodiversity (\ln) index. It ranges from -6.7 to +4.1.

Considering that a “no change” pixel is given a value of 1, the indexes are then expressed as $(\text{idx} + 100)/100$. They range then from 0.93 (degradation) to 1.04 (improvement).

Measurements of species population “favourable” (decrease or stability) or “unfavourable” (decrease) express a trend. “Future prospect” good or unfavourable is of course a judgement of future trend. However, the index is not a measurement of any biodiversity stock. The 2006 Biodiversity index is therefore interpreted as a trend index ranging from 0.93 to 1.04 (or - 7% to + 4%). This normalisation can be refined later on. As such, it means that biodiversity decreasing at an annual rate of e.g. 3% will come close to 0 in one century. It would, in the case on an increase of 3%, be multiplied by 20.

Estimation of series from the past (2005 to 2000) and the future (2007 to 2016)

The ln indexes converted to % are used for estimating the trend. For 2006, the assessment relied on several variables (e.g. land cover, conservation status of species, and habitats), and finally on the sum of POP and FUT indexes. To assess past trends, we used only the POP index, and for future trends the FUT index. The assumption is that the rate of decrease from 2006 to 2000 (POP) is constant, also a constant rate applies to the period 2006 to 2016 (FUT)..

To estimate the index in the past from 2000 to 2005 two scenarios are considered, population unfavourable or stable.

To estimate the index from 2007 to 2016, three scenarios are envisaged: population unfavourable, stable or improving.

The annual index was calculated in SAGA-GIS calculator as:

$$BDV_{year_idx} = BDV_{2006_idx} \times \left(\frac{BDV_{popbad_idx}}{BDV_{fut_all_idx}} \right)^{year-2006}$$

$$BDV_{popbad_idx} = \frac{(100 + POP_ln_idx)}{100} \quad BDV_{fut_all_idx} = \frac{(100 + FUT_ln_idx)}{100}$$

On this basis, the expected outcomes are as follows:

If ..._ln_idx < 0, biodiversity degradation occurs with time, indicated by BDV...idx < 1.

If POP_ln_idx = 0, biodiversity condition is stable between 2000 and 2005, and BDVpop_idx = 1.

If additionally FUT_ln_idx = 0, biodiversity condition is stable between 2007 and 2016, and BDVfut_all_idx = 1.

If POP_ln_idx < 0 but FUT_ln_idx > 0, the biodiversity condition is improving from 2007-2016, and BDVfut_all_idx = 1.

The simulations are illustrated in **Figure 5.12**.

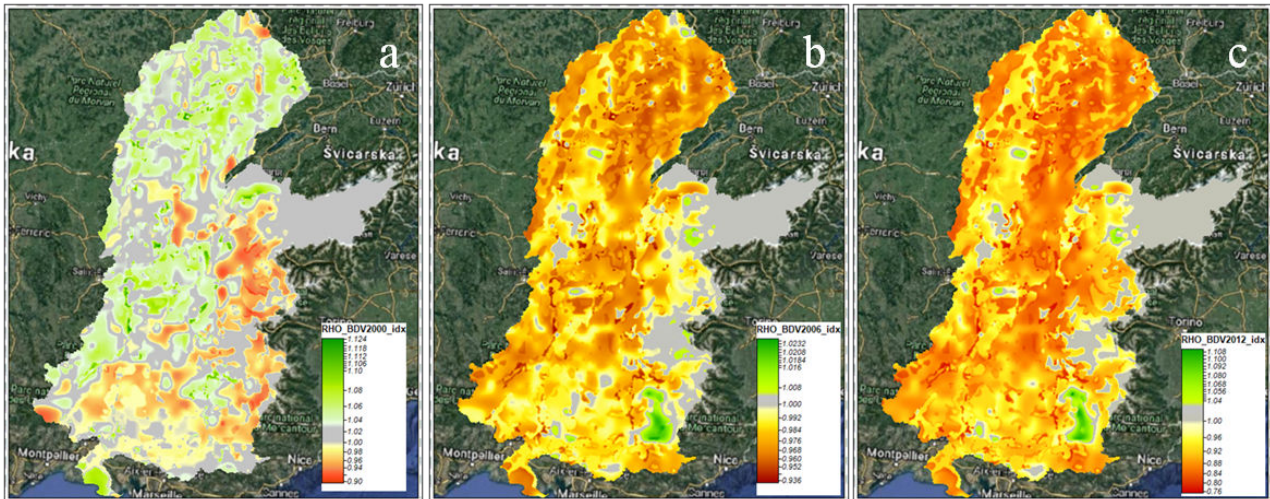


Figure 5.12. Biodiversity trend index for 2000 (a), 2006 (b) and 2012 (c) estimated from Article 17 report. Values larger than 1 (in green) reflect that positive expert's observations outdo negative ones, while values smaller than 1 show degradation. The maps illustrate a widespread and continuous degradation future trend for biodiversity with few exceptions in the Alps.

5.3 Results and Discussion

The Ecosystem Infrastructure account is a promising analytical approach and tool of the ENCA methodological system. It has combined for the first time a genuine series of existing and novel synthetic indicators. Their integration deserves further exploration, but the exercise is first of all tributary of data sources required to generate higher quality results within the ENCA core accounts. In this context, DLCTs constitute a relevant and constitutive component and foundation.

5.3.1 Basic Balances and Accessible Ecosystem Potential

The columns in the table of the Ecosystem infrastructure Functional Services display the averages by Dominant Land Cover Type and the average values for the Rhône watershed (**Table 5.1**). This is a compact version of the genuine accounting tables from the ENCA-QSP (Weber, 2014).

Table 5.3. Ecosystem Infrastructure Functional Services balance sheet compact version.

Ecosystem Infrastructure Functional Services Accounts

<i>Ecosystem Accounting Unit Types</i>	Socio-Ecological Landscape Units (SELU) / Dominant Land Cover Type (DLCT)										landscape ecosystems
	<i>Artificial areas</i>	<i>Agricultural land</i>	<i>Pastures</i>	<i>Agricultural mosaics</i>	<i>Forest</i>	<i>Shrubland and heathland</i>	<i>Natural grassland</i>	<i>Open spaces</i>	<i>Glaciers</i>	<i>Wetlands and water bodies</i>	
I. Basic balances											
II. Accessible ecosystem infrastructure potential											
NLEP 2000 Net Landscape Ecosystem Potential	0.0832	0.3246	0.3627	0.3413	0.5361	0.4721	0.4809	0.6559	0.5763	0.6324	0.4466
NREP 2000 Net River Ecosystem Potential	0.0117	0.0091	0.0092	0.0106	0.0121	0.0167	0.0224	0.0198	0.0193	0.0080	0.0139
TEIP 2000 Total ecosystem infrastructure potential	0.0567	0.1682	0.1860	0.1766	0.2716	0.2441	0.2509	0.3345	0.2979	0.3088	0.2295
NLEP 2006 Net Landscape Ecosystem Potential	0.0829	0.3250	0.3606	0.3381	0.5350	0.4723	0.4807	0.6564	0.5763	0.6326	0.4460
NREP 2006 Net River Ecosystem Potential = RS2 x REP_idx	0.0121	0.0093	0.0091	0.0108	0.0124	0.0177	0.0227	0.0209	0.0203	0.0081	0.0143
TEIP 2006 Closing stock of ecosystem infrastructure potential	0.0567	0.1685	0.1849	0.1751	0.2711	0.2448	0.2509	0.3353	0.2984	0.3090	0.2295
NLEP 2012 Net Landscape Ecosystem Potential	0.0828	0.3243	0.3600	0.3370	0.5348	0.4723	0.4806	0.6565	0.5763	0.6324	0.4457
NREP 2012 Net River Ecosystem Potential	0.0126	0.0096	0.0089	0.0109	0.0126	0.0186	0.0230	0.0218	0.0212	0.0081	0.0147
TEIP 2012 Total ecosystem infrastructure potential	0.0569	0.1684	0.1846	0.1747	0.2711	0.2452	0.2510	0.3358	0.2989	0.3089	0.2295
IV. Table of indexes of intensity of use and ecosystem health											
I_IU 2000 Ecosystem infrastructure use intensity	0.5000	0.5002	0.4995	0.4993	0.4997	0.5003	0.5000	0.5004	0.5002	0.5001	0.5000
I_IU 2006 Ecosystem infrastructure use intensity	0.5001	0.5000	0.4996	0.4995	0.4999	0.5003	0.5000	0.5003	0.5002	0.5000	0.5000
I_IU 2012 Ecosystem infrastructure use intensity	0.5001	0.4999	0.4998	0.4998	0.5000	0.5002	0.5000	0.5003	0.5002	0.5000	0.5000
I_EH 2000 Composite ecosystem health index	1.0157	1.0151	1.0259	1.0203	1.0144	0.9801	1.0023	0.9848	0.9995	1.0549	1.0113
I_EH 2006 Composite ecosystem health index	0.9877	0.9836	0.9798	0.9816	0.9779	0.9873	0.9830	0.9956	0.9997	0.9721	0.9848
I_EH 2012 Composite ecosystem health index	0.9455	0.9203	0.9091	0.9137	0.8875	0.9075	0.9023	0.9584	0.9975	0.8893	0.9231

Chapter 5 Ecosystem Infrastructure Accounts

The results of the single indexes used to calculate the NLEP are presented in **Figure 5.13**. The highest values of Green Background Landscape Index correspond to the less artificial Dominant Land Cover Types: Open Spaces, Glaciers, and Wetlands and Water Bodies. The surfaces with the highest probability of connecting two points randomly chosen correspond to smaller fragmentation areas, namely Forest, Open Spaces, and Wetlands and Water Bodies.

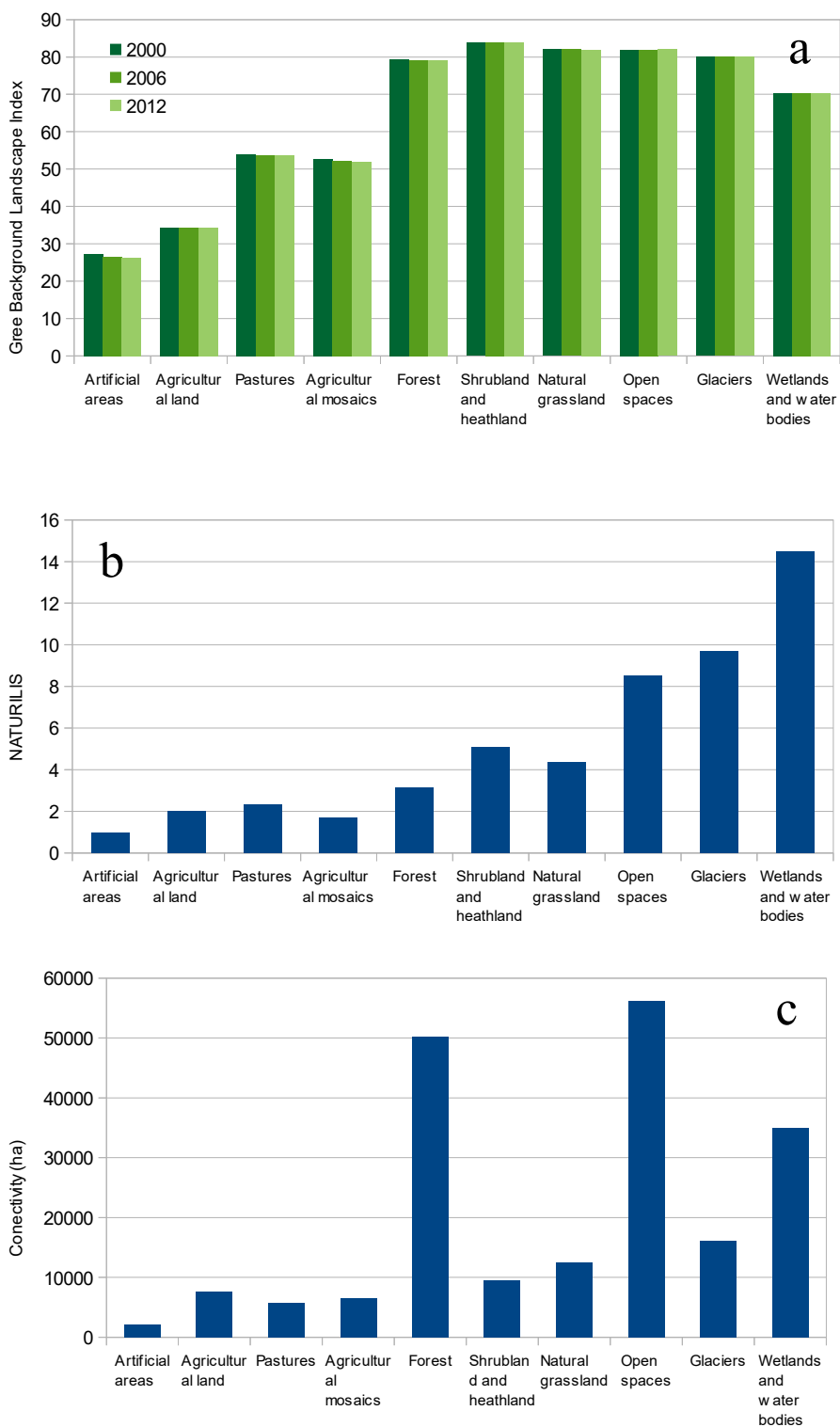


Figure 5.13. Green Background Landscape Index (a; Figure 5.4), NATURILIS (b; Figure 5.5), and CONECTIVITY (c; Figure 5.6) by Dominant Land Cover Type (Figure 3.13).

Chapter 5 Ecosystem Infrastructure Accounts

For NREP calculation the two-component index, Rivers Condition Potential and River Natural Conservation Value, is shown by Dominant Land Cover Types in **Figure 5.14**. The Rivers Condition Potential is higher in Natural Grasslands, Open Spaces, and Glaciers. The rivers in protected areas show the same pattern as NATURILIS.

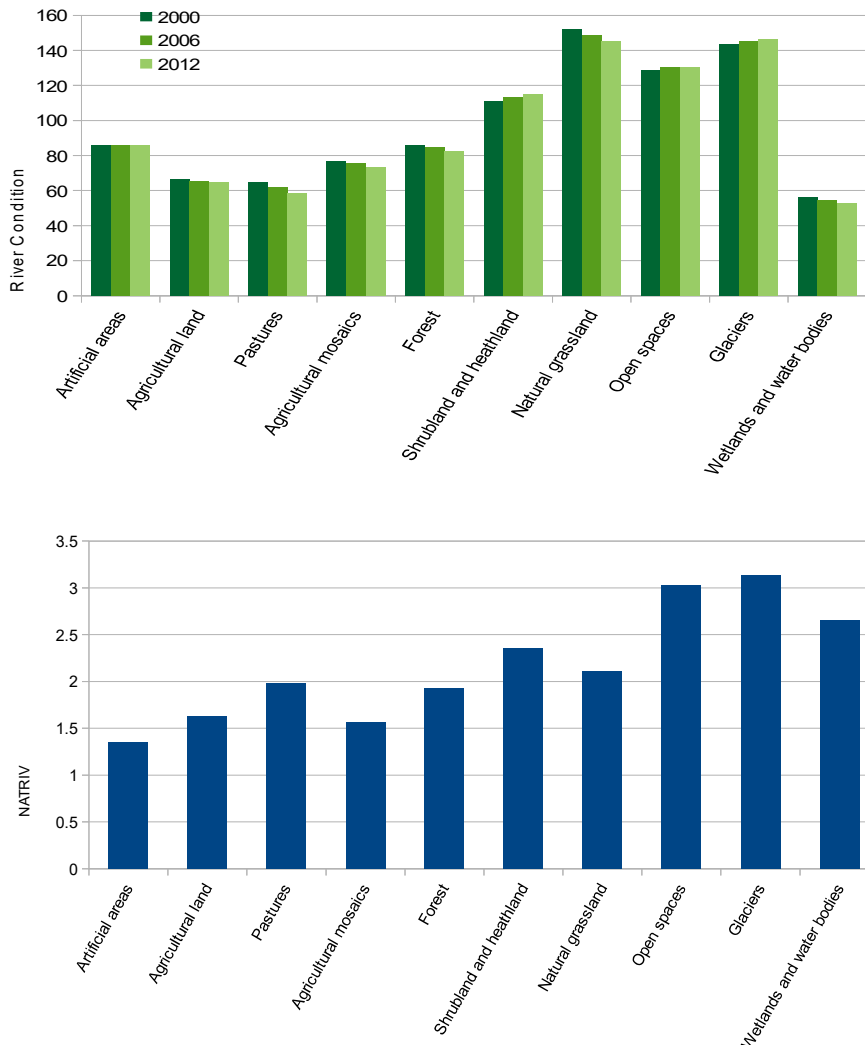


Figure 5.14. Rivers Condition Potential (top; Figure 5.7) and River Natural Conservation Value (NATRIV; Figure 5.9) by Dominant Land Cover Type.

As one can see from the above results the changes within each Land Cover Type category over the entire 2000-2012 period are small for the reported indicators.

5.3.2 Total Ecosystem Infrastructure Potential

The integration of the NLEP and NREP shows that Open Spaces, Glaciers, and Wetlands and Water Bodies, followed by the Forested cover, have the highest Ecosystem Infrastructure Potential (Figure 5.15).

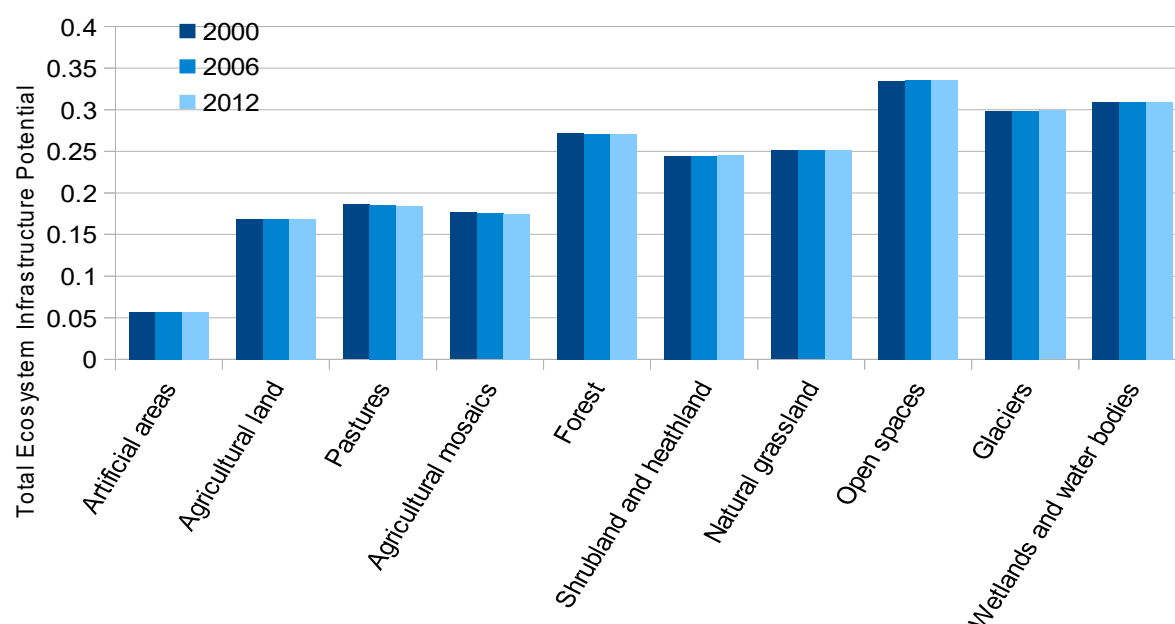


Figure 5.15. Total Ecosystem Infrastructure Potential (Figure 5.10) by Dominant Land Cover Type (Figure 3.13).

5.3.3 Indexes of Intensity of Use and Ecosystem Health

The *Intensity of Use* values relatively closely reflect the pattern of Dominant Land Cover Types (DLCT) with their corresponding Total Ecosystem Infrastructure Potential (Figure 5.16). Interestingly, the Intensity of Use shows that the Ecosystem Infrastructure Potential is mainly diminishing in man-made landscapes, with the notable exception of “agricultural land” *per se*. This is somehow paradoxical, and deserves further investigation. One can speculate that this is because agricultural land is an extreme sort of landscape that has reached a glass ceiling in terms of potential and intensity of use. These two aspects should be analysed at various scales and cross-checked with the contribution of concerned stakeholders and actors.

The *Ecosystem Health* index evaluation shows improvement in changes (values < 1) in most of the DLCT in 2000 (Table 5.3, row I_EH 2000), in 2006 and 2012 only degradation was observed (values > 1) in all DLCT (Table 5.3, rows I_EH 2006 and 2012).

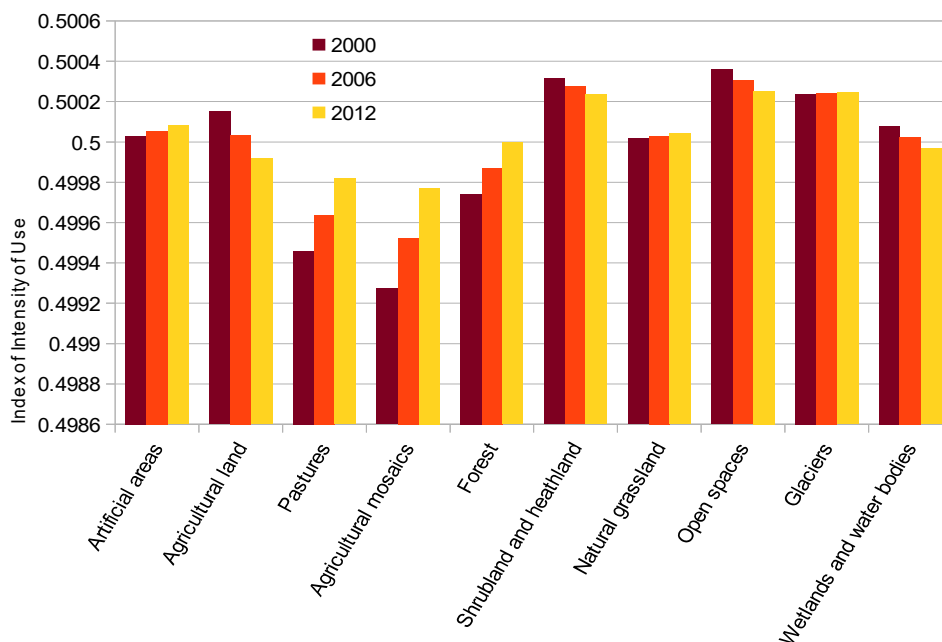


Figure 5.16. Index of Intensity of Use (Figure 5.11) by Dominant Land Cover (Figure 3.13).

As a matter of fact, the integration of a Biodiversity index in whatever environmental evaluation approach is likely to be a challenge (see for example the planetary boundary context, Steffen et al, 2015 and the **General Introduction**). In the case of ENCA, a first option has been using the methodology based on Article 17 reporting in calculating the Ecosystem Health Index. Alternative methodologies need to be explored in the future (see below). However, the ENCA approach to the evaluation of Ecosystem Infrastructure integrates an important landscape dimension, and the corresponding set of indicators. As such, they constitute a major Essential Biodiversity Variable (EBV; discussed in the General Introduction) that we have taken into account.

As a first level of clarification, we compared the spatial patterns of the reference state of NLEP, NREP, TEIP, and Biodiversity index, as shown in **Figure 5.17**.

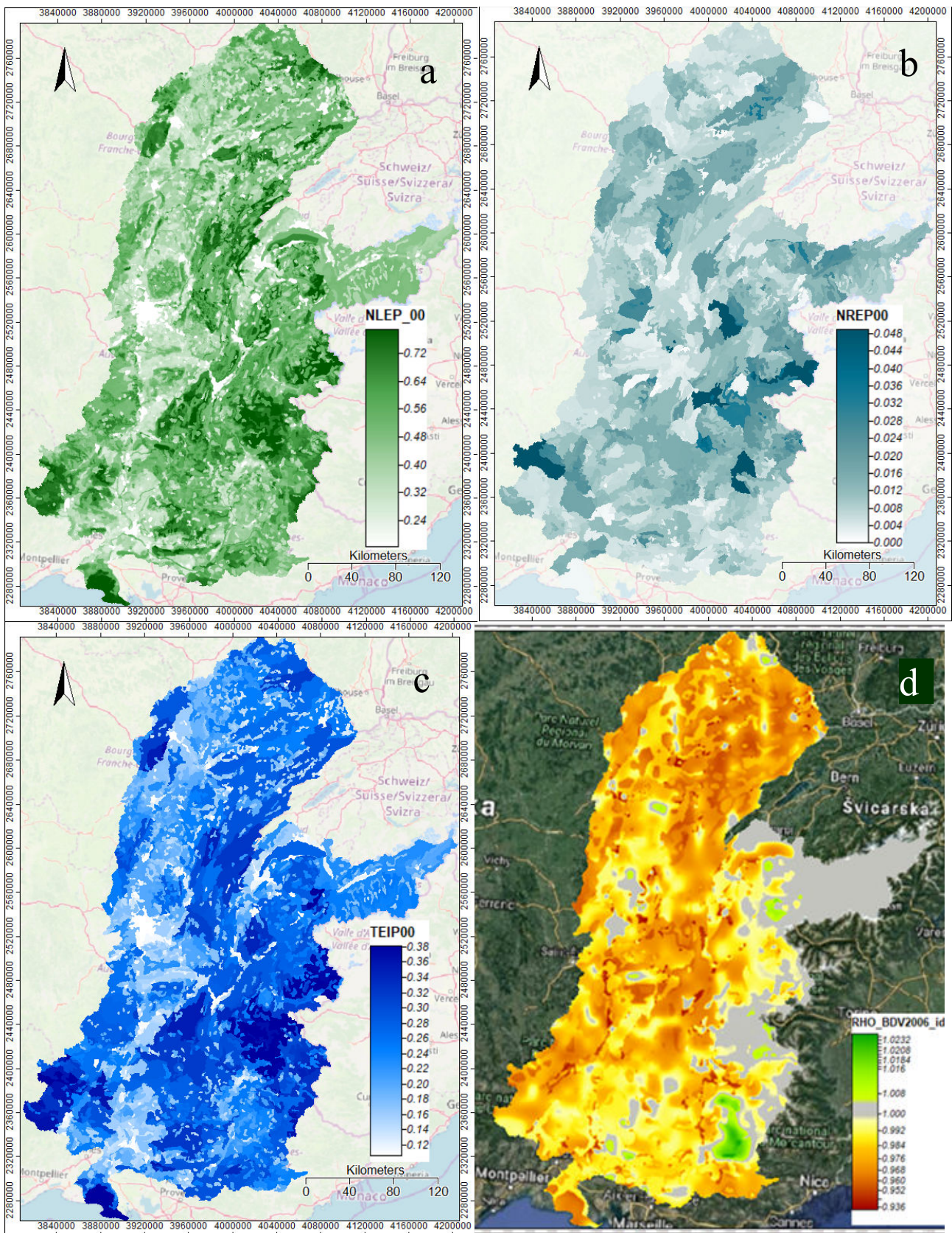


Figure 5.17. Comparison of NLEP (a; Figure 5.3), NREP (b; Figure 5.7), TEIP (c; Figure 5.10), and Biodiversity index (d; Figure 5.12).

Chapter 5 Ecosystem Infrastructure Accounts

The Figure allows to visualize the patterns generated by each of the four synthetic indicators with respect to their geographic distribution according to Dominant Land Cover Types (cf. **Figure 5.16**). The Figure also shows that the landscape indicators, and NLEP in particular, are likely to acceptably describe the state and dynamics of the ecosystem infrastructure, a structuring condition of the ecosystem capital. As a matter of fact, the data sources required for generating these landscape indicators are at this point the most robust and relevant.

As a second level of clarification, we compared the relative patterns of distribution of natural areas (ONB, 2017; **Figure 5.18a** with reference to **Figure 5.19b** representing the effective mesh size by SELU) and the gradient of pressure on biodiversity (ONB, 2019; **Figure 5.18c**) with the ENCA Rhône Biodiversity index (**Figure 5.18d**).

The distribution pattern of the mesh effective size of natural areas by forestry region by department in France in 2006 (ONB, 2017) and our Socio-Ecological Landscape Units in 2000 (despite the difference in their respective mesh size, respectively 99,97 km² for France and 330.62 km² per SELU), show relatively similar profiles. This is also the case with the patterns of the gradients of pressures on biodiversity (namely soil artificialization, resource overexploitation, climate change pollution, and invasive alien species; OBN, 2017). In all these cases the state and pressures on ecosystems are used to evaluate biodiversity sensu Article 17 report from the European Topic Center on Biological Diversity (ETC-BD, Section 5.3, Index of Ecosystem Health).

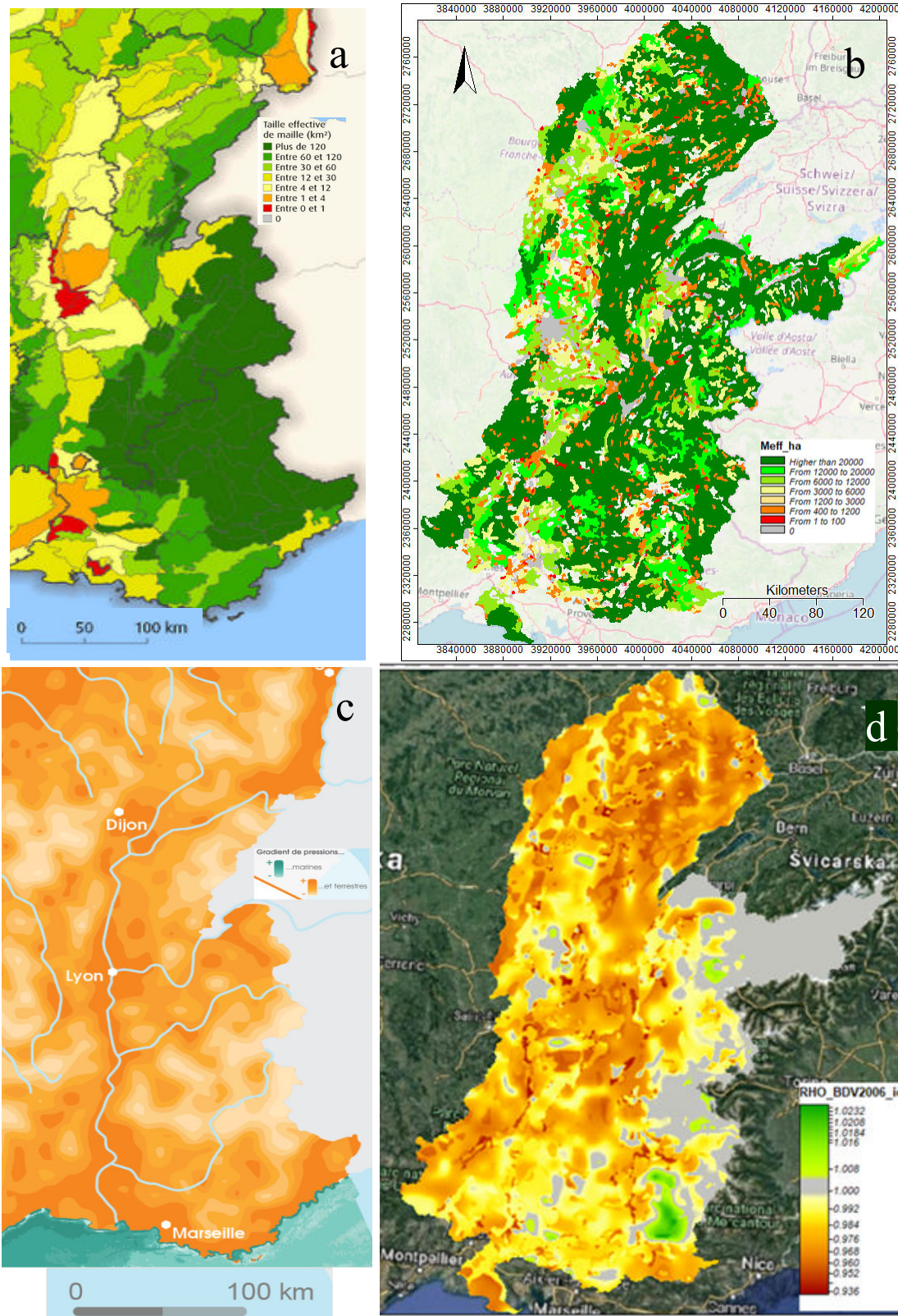


Figure 5.18. Comparative representations of Biodiversity evaluation. Mesh effective size of natural areas by departmental forestry region (a) from the *Observatoire National de la Biodiversité* (OBN, 2017), Mesh effective size of ENCA Rhône watershed by SELUs (b; Figure 5.6), gradients

Chapter 5 Ecosystem Infrastructure Accounts

of pressures on biodiversity (c) OBN (2017), and Biodiversity Index ENCA Rhône watershed (d; Figure 5.12b).

Of note, the Figure shows the relative coherence of these representations of Biodiversity, with the remarkable resolution of the ENCA/SELU results.

Taken together, the above observations suggest that the landscape / habitat parameters are apparently an accurate and sufficient level of diagnostic to evaluate trends in biodiversity. A notable advantage making it possible to simultaneously monitor changes in biodiversity world-wide with a reasonable periodicity.

With these considerations in mind, it is important to note that these two macro indexes of the Ecosystem Infrastructure account, Intensity of Use and Ecosystem Health, are going to be integrated for the calculation of the Ecosystem Capability Unit (see **Figure 5.2 and Chapter 6**).

5.4 Main issues

NLEP Net Landscape Ecosystem Potential

The scores of Landscape high nature conservation value index NATURILIS have been elaborated according to the French management categories. In other countries, the evaluation system depends on the corresponding environmental legislation and the protected areas categories. Another issue is that no change cannot be evaluated because the surface of most of the designated areas does not change from year to year. This index was calculated only for the French part of the watershed.

Concerning the Landscape Connectivity index, the indicator only counts the surface of the effective fragmentation area, without considering the barrier effects it generates and its ecological implications.

NREP Net River Ecosystem Potential

To calculate the srkm a semi-statistical model was used to determine the average runoff of river classes by hydrological units.

For NATRIV, the concerns are similar to those mentioned for NATURILIS.

Indexes of Intensity of Use and Ecosystem Health

With respect to the Index of Intensity of Use, the change has been evaluated for only one of the three indexes due to the lack of time series.

Concerning the Biodiversity Index, our estimations are based on the 2006 benchmark. Annual estimations are produced by expert judgements of population and future prospects as one time point with a range of 0,11. In the EEA experimental approach to ecosystem capital accounting, linkage between species data and land was the focus. The ecological niche modelling (Peterson et al, 2011) was considered highly relevant, as it is based on land and species time series. However, its implementation at the continental scale requires additional developments: while land cover data in Europe seem fit for the approach, more data on species are required. Other solution is using the communities specialism index (CSI) which correlates change in species composition and ecosystem condition (Loïs et al 2009; Devictor, 2007). The formula is a ratio of specialist species to non-specialist species. High CSI value report a good condition for biodiversity considering both species and ecosystems. As opposed to the so-called “richness” indicators based on counts of species, CSI values those specialist species which are generally endemic or representative of the evaluated area. CSI is established by broad habitat types corresponding to land cover classes and derived dominant land cover types as defined in ENCA. With appropriate spatial details, it could be usefully

Chapter 5 Ecosystem Infrastructure Accounts

implemented in ENCA. CSI has been developed in France and in other countries. In France, application on birds has proved to be highly relevant.

Conclusion

In this Chapter two intermediate macro indicators on ecosystem capacity to deliver functional services were calculated, namely NLEP and NREP. They revealed a moderate NLEP (0.45) and a very low NREP (0.01) (cf. **Figure 5.7**). The integration of the two indicators resulted in the Total Ecosystem Infrastructure Potential (TEIP). The Intensity of use was calculated as the change of the TEIP between two time periods, with as result a moderate value (0.5). The Biodiversity index showed improvement (>1) in 2000 and degradation (<1) in 2006 and 2012. These indexes will serve to the calculation of infrastructure unit value and its integration to the Ecosystem Capability Unit in **Chapter 6**.

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5.5.1 Data sources

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ONEMA/IRSTEA Available from <https://webgr.irstea.fr/en/research-2/discharge-mapping/>

Chapter 6 Integration of Accounts: The Ecosystem Capital Capability Account

Outline

In this chapter I calculate the Unit Values of each of the three accounts, namely bio-carbon, water, and ecosystem infrastructure. On the basis of the geometric mean of the Intensity of use and the Index of health we produce a single synthetic macro-indicator of the ecological value, the Ecosystem Capital Capability (or overall Potential). As composite figure, the ECU is at the apex of the ENCA approach. Changes of ECU values over time are reported. They allow evaluating trends that inform on the level and rate of ecosystem capital degradation or enhancement.

6.1 Introduction

In physical accounts, measurements are made in basic units (tons, joules, m³ or ha) and converted to a special composite currency named ECU for 'Ecosystem Capability Unit'. The price of one physical unit (e.g. 1 ton of biomass) in ECU expresses at the same time the intensity of use of the resource in terms of maximum sustainable yield and the direct and indirect impacts on ecosystem condition (e.g. contamination through pollution or biodiversity loss). Loss of ecosystem capability in ECU is a measurement of ecological debt. To territorial debt, it should be added the consumption of non-paid ecosystem capital that is embedded in international transactions. The ecological debt in ECU (and symmetrically credits when improvements are verified) could be incorporated into portfolios of financial instruments. Physical degradation or ecological debts can be recovered in a second time into euros on the basis of the costs necessary to restore ecosystems capability (Weber, 2012).

The ENCA methodology (Weber, 2014) for calculating Ecosystem Capability by means of the Ecosystem Capability Unit (ECU) was performed. The composite Index based on Bio-carbon, Water, and Infrastructure indexes measures the “price” corresponding to the ecological value of the ecosystem capability (see **Box 5.1**).

Because “prices” in *ECU integrate intensity of resource use and other ecosystem condition indicators*, a change in “price” of the accessible resource in ECU will capture the degradation or conversely the development of the ecosystem capability to deliver services in a sustainable way. From the documented information and indexes extracted to SELUs from previous chapters, the ECU values were ultimately calculated (Figure 6.1).

Accounts for an ecosystem unit

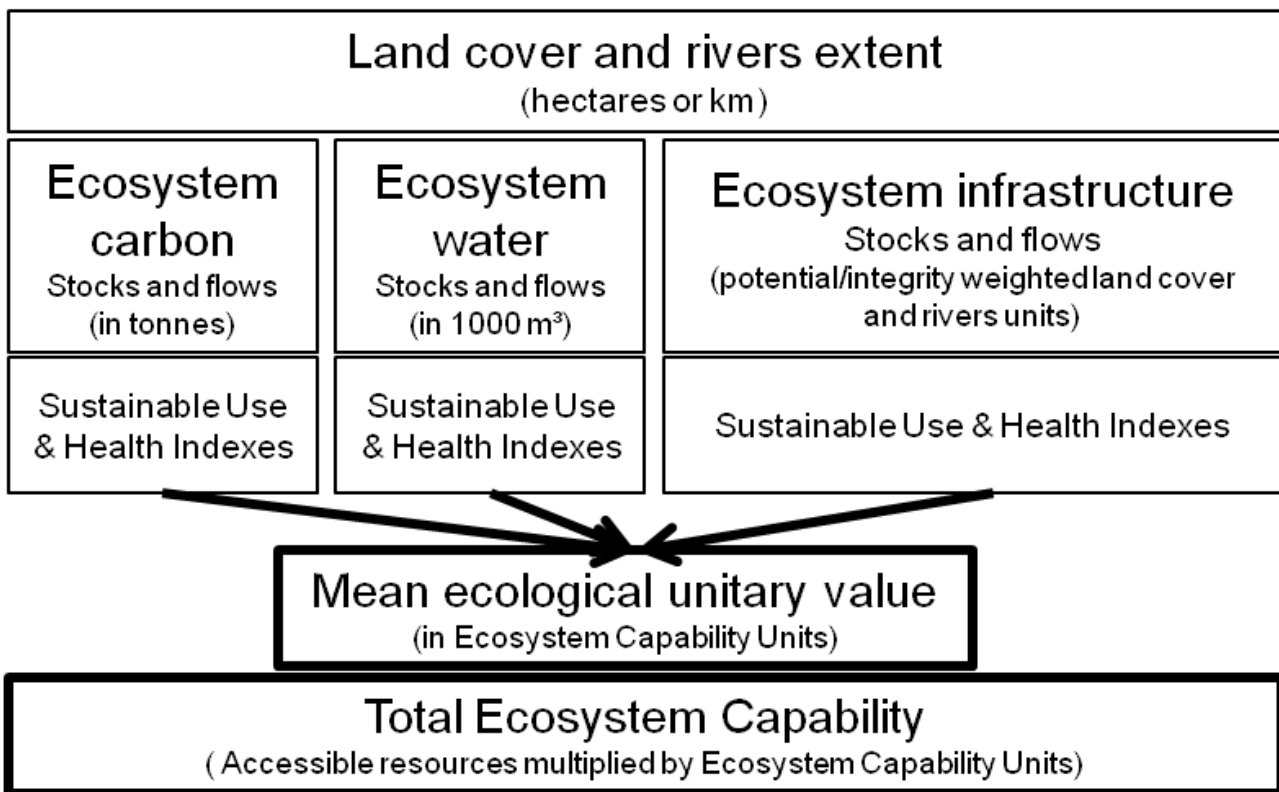


Figure 6.1. Articulation and integration of ENCA components. This Chapter addresses The Ecosystem Capability, shown in bold, and integrates the previous synthetic indicators.

For evaluating the Intensity of Use, the ecosystem water accounts are a demonstrative case. To capture the key aspects of natural water flows and their abstraction and transportation for human needs, and the solidarities inside the hydrological systems, the ambition to derive indexes of **sustainable intensity of use** directly from water balances cannot be met. The ENCA methodology foresees such situations and recommends in this case to use diagnoses instead of strict quantitative balances. The hydrologists came, since long, to the same conclusion and have developed best practice procedures accordingly.

The water **ecosystem health** issue is a good example (also see **Chapter 4**). The diagnosis approach recommended by ENCA is based on Health indicators, such as:

- Bio-chemical quality diagnosis for all water categories; in the case of rivers the diagnostic is based on srkm_Water measurements used to weight the DCE indicator of “good” (1) or “bad” (2) chemical status);
- Vulnerability to pollution, such as nitrates, addressed using the reporting to the Nitrates Directive of the EU and recorded;
- Dependence on artificial inputs, such as agriculture dependence to irrigation measured by the ratio of green water to irrigation water and recorded.

In what follows, I reproduce the ECU methodology in the context of the Rhône river basin. Seen the composite nature of the ECU and its origination through a series of building blocks that have their own technical shortcomings, the exercise is essentially a blind test. However, getting at the end of the ENCA Rhône road is a necessary development of this work, meant to give a first level understanding of underlying problems and ways to address them.

6.2 Methods

Calculation of Mean Internal Unit Values

According to the ENCA methodology, the **index of sustainable intensity of use** measures the excess of use resulting in resource depletion (see **Figure 6.1**). For carbon, the index is defined as the accessible resource / total use ratio. For water, the index is produced through diagnosis of the quantitative status. The respective quantifications are normalized to range from 0 to 1. Thus, values of 0.5 signify stable, > 0.5 improvement, and < 0.5 degradation. For the ecosystem infrastructure, it is the change between two dates that defines the index: the value of 0.5 means stable, > 0.5 improvement, and < 0.5 degradation.

The indexes of intensity of use and ecosystem health of each account, namely bio-carbon ecosystem account (Chapter 3), water ecosystem account (Chapter 4), and infrastructure ecosystem account (Chapter 5) are combined as the geometric mean to calculate their respective unit values per SELU. The mean of pixels by SELU was extracted using SAGA-GIS *Grid Statistics by Polygons* and the geometric mean was calculated with QGIS Field Calculator. The results were rasterized in SAGA-GIS. (**Figures 6.2, 6.3, and 6.4**).

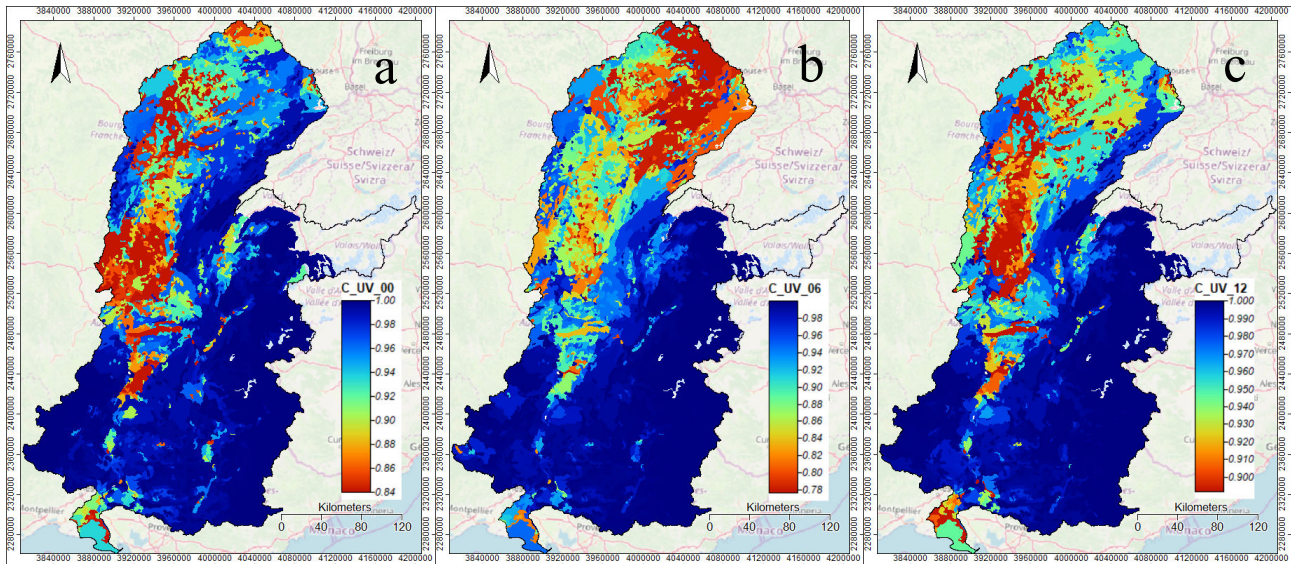


Figure 6.2. Carbon Account Mean Internal Unit Value by SELU, 2000 (a), 2006 (b), and 2012 (c). This is calculated as the geometric mean of the Index of Intensity of Use (Figure 3.16) and a default value of Ecosystem health of 1 using SAGA-GIS geoprocessing tools. The theoretical range is from 0 to 1 (darkest blue).

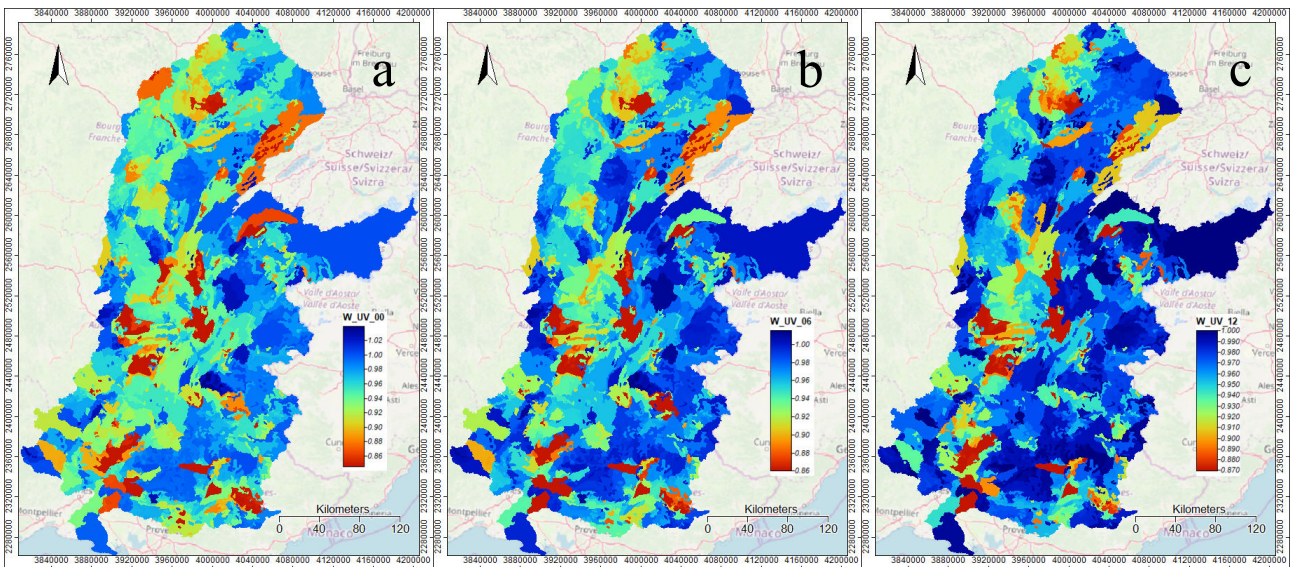


Figure 6.3. Water Account Mean Internal Unit Value by SELU, 2000 (a), 2006 (b), and 2012 (c). This is calculated as the product of the Index of Intensity of Use and an Ecosystem Health Index by ZHYD using SAGA-GIS geoprocessing tools. The theoretical range is from 0 to 1 (darkest blue)

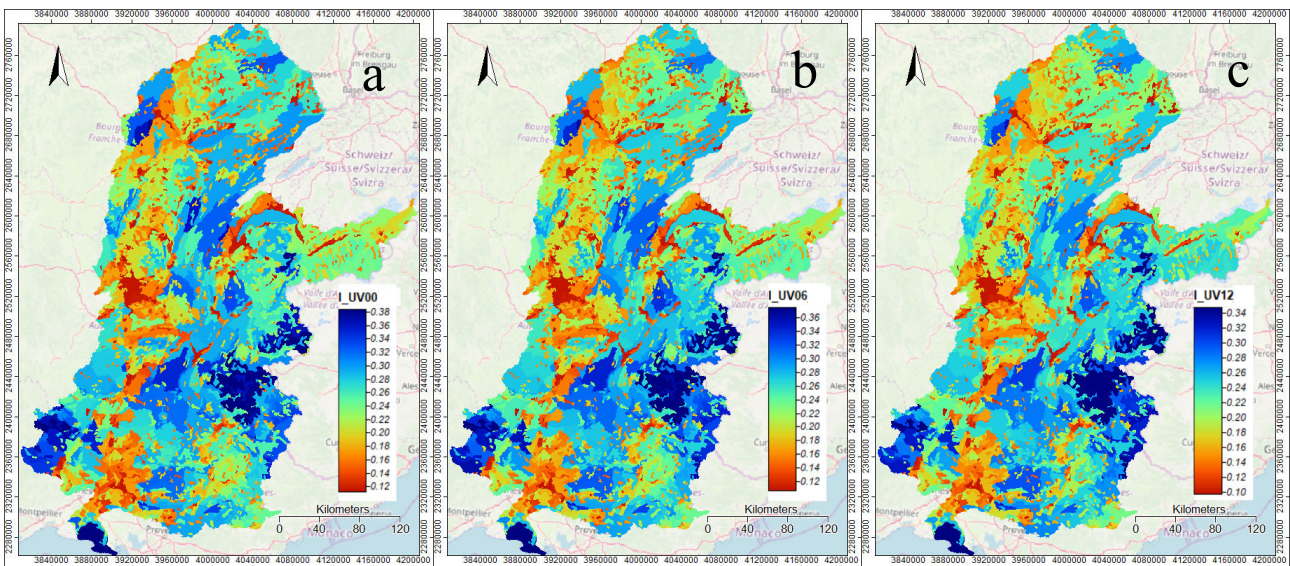


Figure 6.4. Infrastructure Account Mean Internal Unit Value by SELU, 2000 (a), 2006 (b), and 2012 (c). This is calculated as the geometric mean of the Index of Intensity of Use (Figure 5.11) and an Ecosystem Health Index (Biodiversity, Figure 5.12) using SAGA-GIS geoprocessing tools. The theoretical range is from 0 to 1 (darkest blue).

Calculation of Ecosystem Capability Unit

The ECU value was calculated as the geometric mean of the three Mean Internal Unit Values, Carbon, Water, and Infrastructure by SELU. Using the formulas 1, 2, and 3 in QGIS for 2000, 2006, and 2012, respectively:

1. $(\text{"WUV_00"} * \text{"C_UV_00"} * \text{"I_UV_00"})^{(1/3)}$
2. $(\text{"WUV_06"} * \text{"C_UV_06"} * \text{"I_UV_06"})^{(1/3)}$
3. $(\text{"WUV_12"} * \text{"C_UV_12"} * \text{"I_UV_12"})^{(1/3)}$

The results were multiplied by 100 in SAGA-GIS calculator to avoid decimals.

The change in percentage of the ECU value was calculated with the formula:

$$\% \text{ increase} = \text{increase} \div \text{original amount} \times 100$$

$$\text{increase} = \text{final amount} - \text{original amount}$$

6.3 Results and Discussion

The highly exploratory nature of the work in this chapter requires prudent interpretation of the results below. The results are therefore presented as evidence for the coming discussions and debates. They need a thorough examination of the concepts and methodological choices in Chapters 4 and 5, and consultations with experts in the corresponding fields.

The mean Carbon Unit Value by DLCT is shown in **Figure 6.5**. The Artificial areas, Agricultural Land, Agricultural Mosaics, and Wetlands and Water Bodies show the same trend, they increase with each reference year. Pastures, Forest, Shrubland and Heathlands, and Natural Grassland show a reduction in 2006. The high mean Carbon Unit Values for Artificial areas could be interpreted as carbon production that is not harvested in contrast to the other four categories where large quantities of carbon are produced and also harvested.

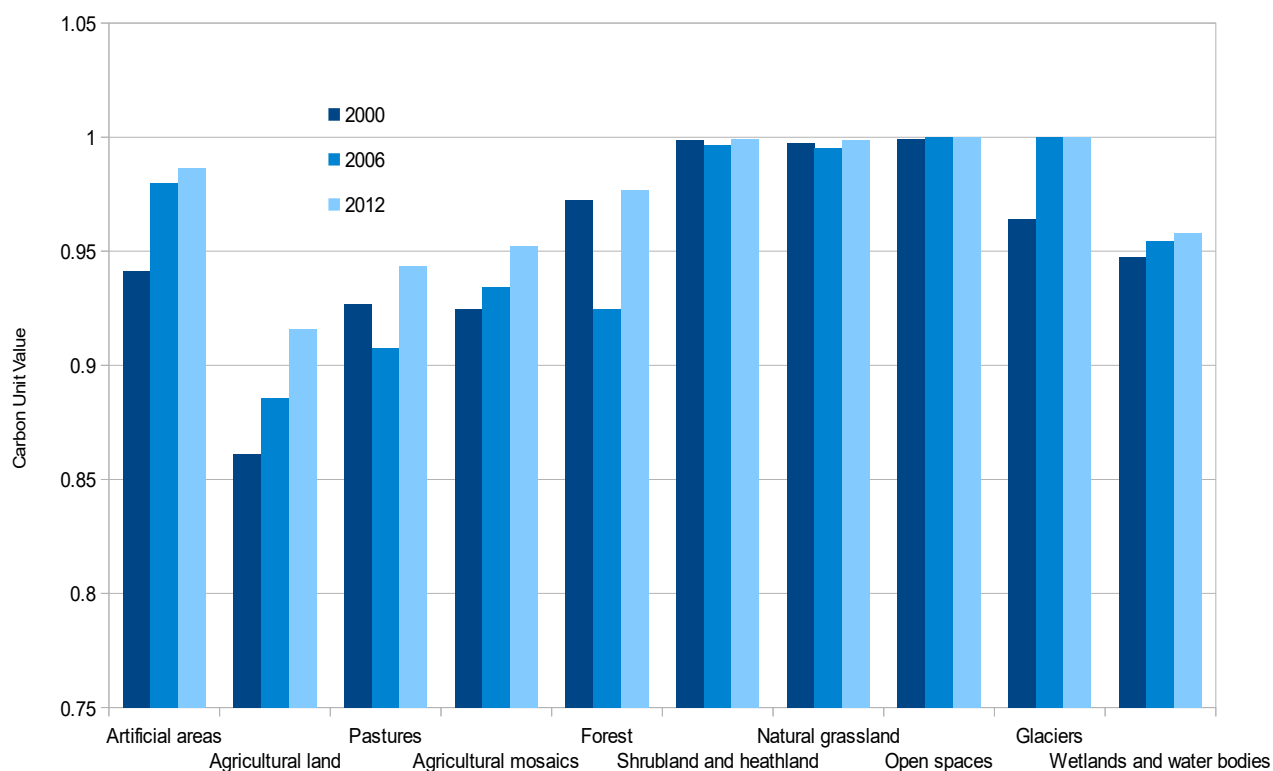


Figure 6.5. Carbon unit value (Figure 6.2) by Dominant Land Cover Type (Figure 3.13).

For the mean Water Unit Value, all categories show an increase in value for the analysed time period, except for the Open spaces and Glaciers that show the opposite trend (Figure 6.6).

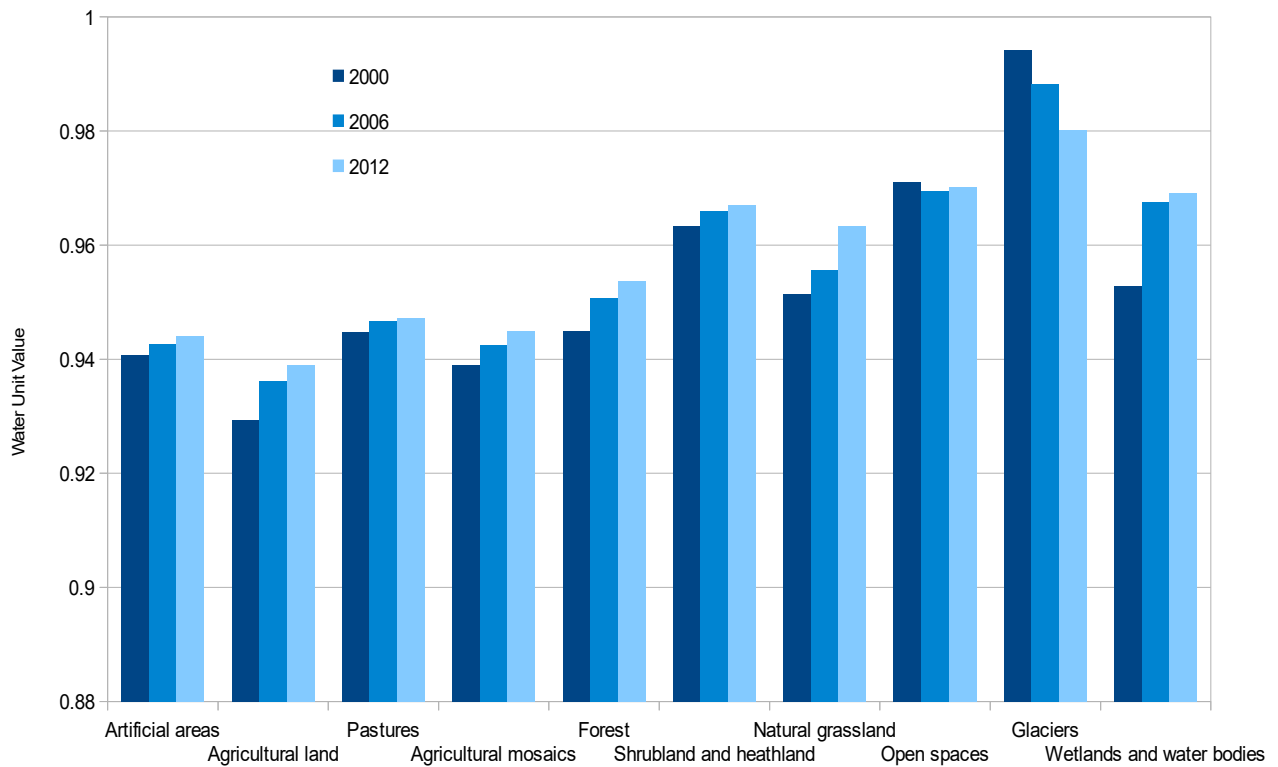


Figure 6.6. Water unit value (Figure 6.3) by Dominant Land Cover Type (Figure 3.13).

The mean Infrastructure Unit Value displays a decreasing profile in most of the DLCTs (Figure 6.7).

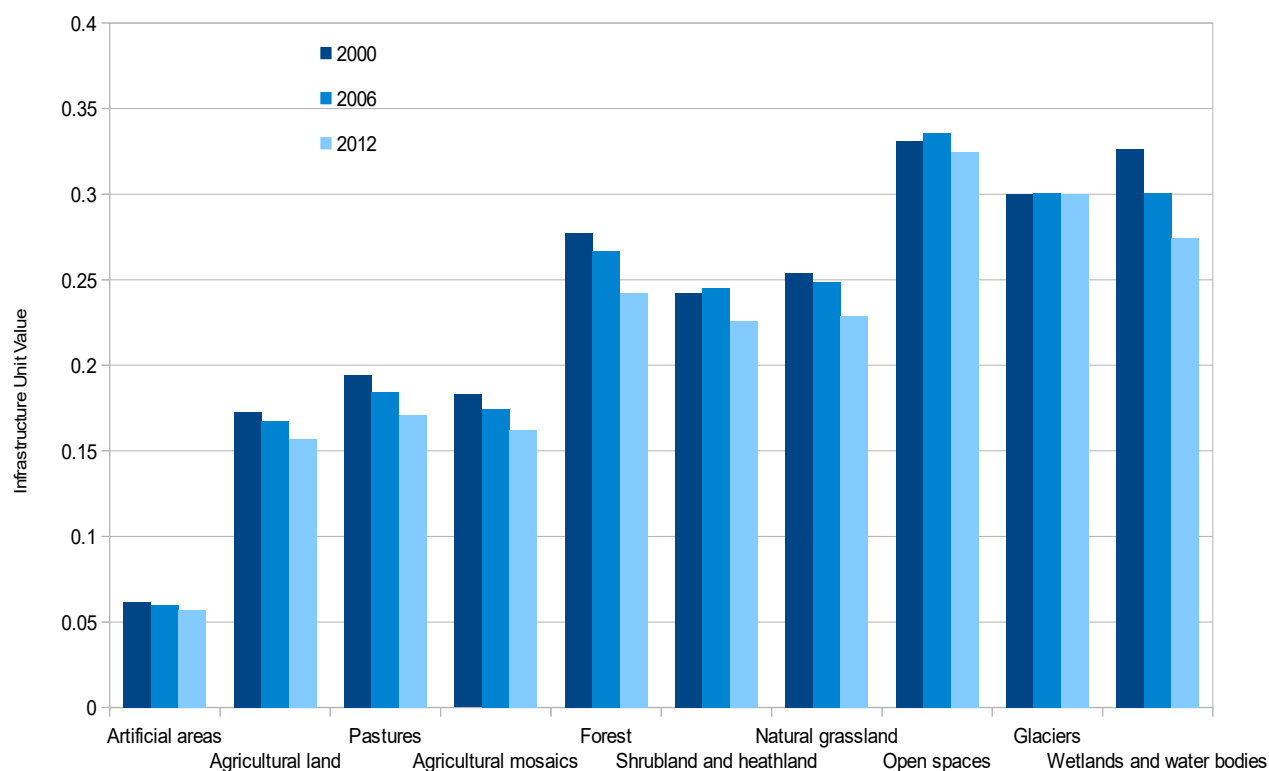


Figure 6.7. Infrastructure unit value (Figure 6.4) by Dominant Land Cover Type (Figure 3.13).

The mean value of the *Ecosystem Capital Capability* in the Rhône watershed for the three years is 59.39 (Table 6.1), which indicates a relatively constant global Ecosystem Capability for the territory during the considered time period.

Table 6.1. Ecosystem capability Unit value and trends for the Rhône river watershed during 2000 to 2012 period.

Year	ECU
2000	60.36 (SD = 7.05)
2006	59.30 (SD = 7.00)
2012	58.51 (SD = 6.68)

The highest ECU values are observed in the east of the watershed (Figure 6.8).

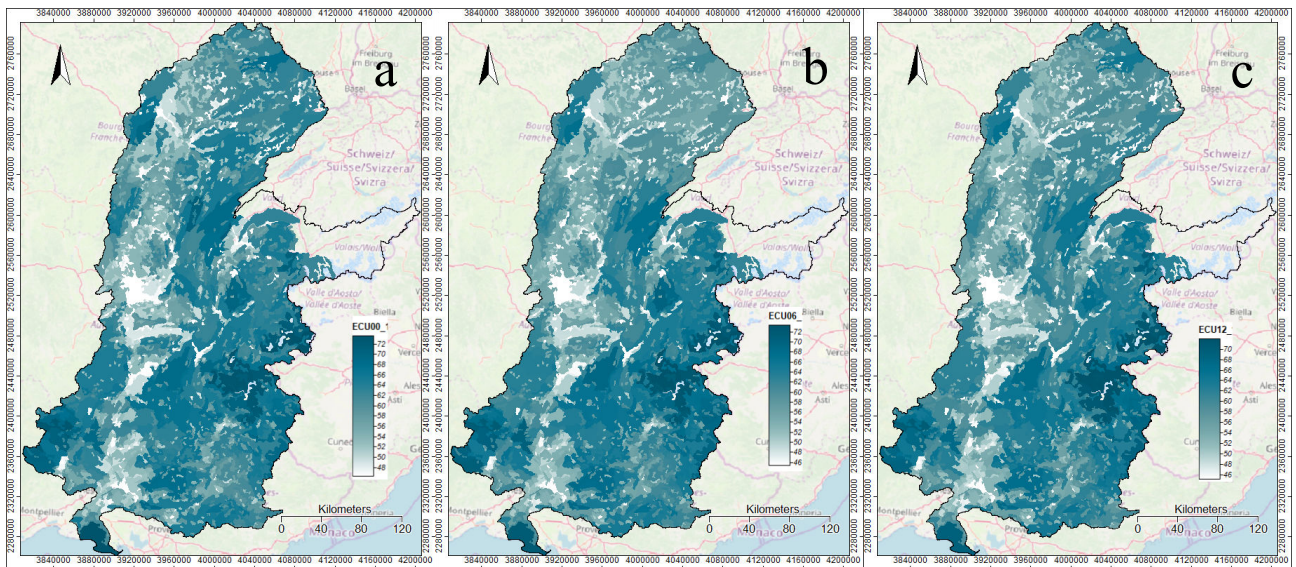


Figure 6.8. ECU value by SELU, 2000 (a), 2006 (b), and 2012 (c). This is calculated as the geometric mean of Carbon (Figure 6.2), Water (Figure 6.3), and Infrastructure (Figure 6.3) Unit Values in SAGA-GIS. The theoretical range is from 0 (white) to 2 (Darkest color).

The mean ECU distribution according to Dominant Land Cover Types is shown in **Figure 6.9**.

The results show a decreasing pattern in most of the categories.

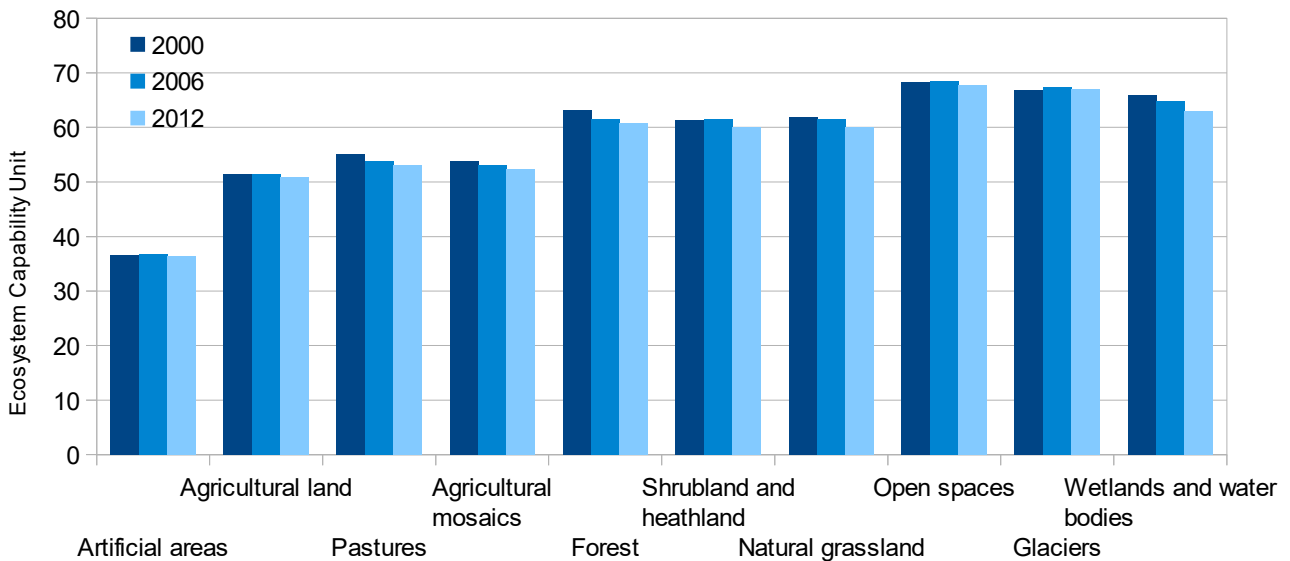


Figure 6.9. Ecosystem Capability Unit (ECU, Figure 6.8) by Dominant Land Cover Type (Figure 3.13).

The negative changes in ECU need to be considered as debt. For the whole period 2000 to 2012, the highest debts (>3%) were to Wetlands and Water Bodies, followed by Forest, and Pastures. For the period 2000 to 2006, the highest debts (>2%) were to Wetland, Shrublands and Heathlands, and Natural Grassland. For the period, 2006 to 2012, the highest debts were to Forest and Pastures (Figure 6.10).

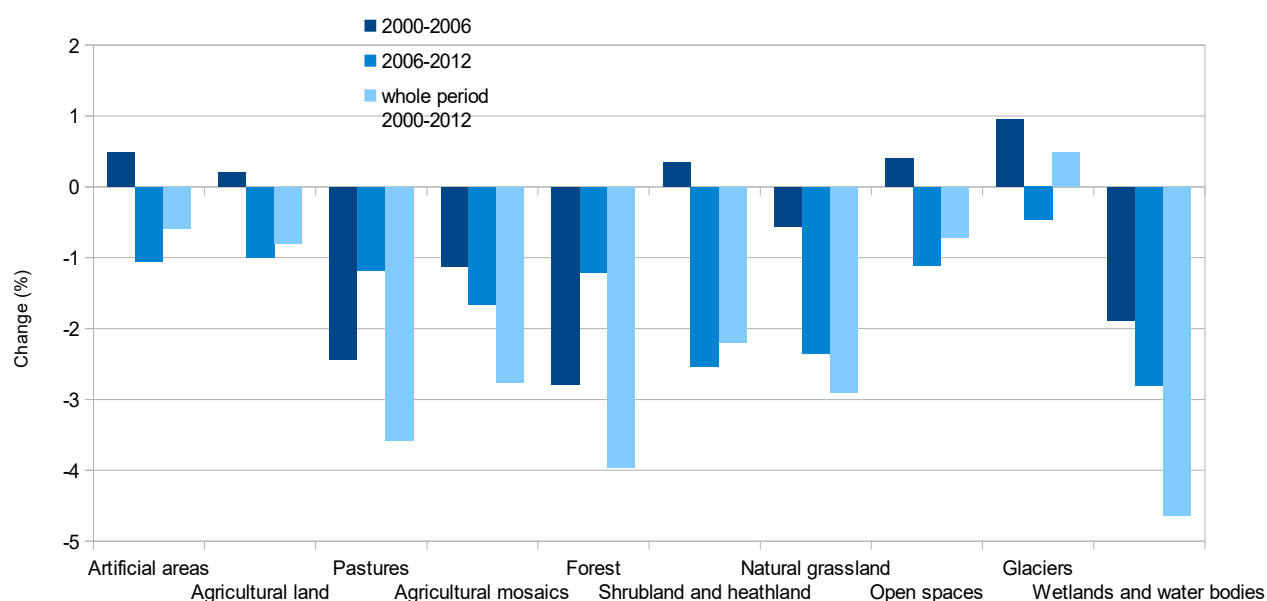


Figure 6.10. Change in percentage in Ecosystem Capability Unit (Figure 6.8) by Dominant Land Cover Type (Figure 3.13).

The mean of change in percentage of the ECU values per hectare for the whole period 2000 to 2012 was -2.98 with SD 2.75, for 2000 to 2006 -1.70 with SD 3.25, and for 2006 to 2012 0.08 with SD 2.24. The spatial distribution of the change in ECU value is shown in Figure 6.11. Of note, in all the series, the SD are high, so they need a cautious consideration.

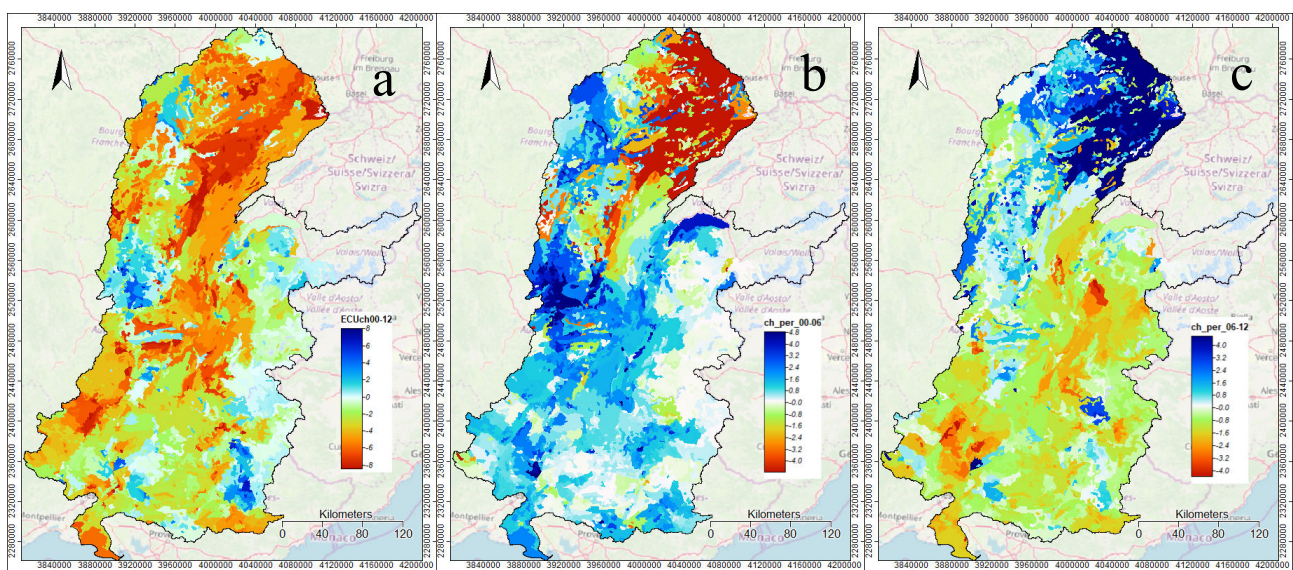


Figure 6.11. Geographic distribution of the changes in ECU values from the whole period 2000 to 2012 (a), 2000 to 2006 (b), and 2006 to 2012 (c) calculated in SAGA-GIS. Blue indicates positive changes, white no changes, and other colors negative changes.

According to the Figure, negative changes (degradation) were mostly observed when the analysis is performed by Dominant Land Cover Type. On the other hand, negative and positive changes were found when the calculations are performed by geographic distribution. The negative changes in the north of the watershed from 2000 to 2006 and its recovery from 2006 to 2012 could be related to the decrease in biomass production in 2006.

In summary, the results reported here are a primer: they are produced largely on an accounting basis, itself largely experimental. From this point of view, the ENCA Rhône test is a significant step forward. There are, however, questions about the results. In general, the DLCT-based statistics for the French part of the Rhône basin strongly mark the results. In a second step, the ECU values should be examined according to the ENCA components in order to better understand the extent to which carbon or water or infrastructure constitute the main explanatory factor(s).

On the issue of data quality, the problem being more general, it can and should be addressed to suggest possible improvements (see Chapter 7, General Discussion). As a matter of fact, what is valid for ENCA also holds for HANPP and for IPCC. For example, we will be able to calculate the NPP from satellite data. The next step should concern data on logging, agricultural statistics, meteorological information, water transfers and distribution, insufficient monitoring of lakes and glaciers. Challenges still remain for soils, which are difficult to observe, but where progress is

made, or for the ecological condition of rivers. The latter is in fact a similar evaluation process to that used for the biodiversity index: it consists in exploiting datasets that are subsequently interpreted by the experts of the field.

6.4 Main issues

For the reasons already exposed in the introduction the issues as identified here need to be cautiously looked into.

We have met two difficulties when working for the ECU implementation. The first one is linked to the quality of input data, which is probably not sufficient for computing reliable indexes and the actionable context of their application. The second is more conceptual, as it questions the notion of accessible resource in accounts based on small spatial units or requires additional developments on the intensity of use for each individual account.

A range of assumptions for each of the accounts have been used to calculate the ECU values.

Comparable scales and normalization of each of the indexes should be considered, which implies larger computational costs. Moreover, special attention should be paid to calculations when results produce zero values, as well as for no data values, to be able to use the result in the subsequent steps of the calculation procedures. Similarly, the necessary inter-calibration of the indices needs particular attention and some adjustments.

Finally, the integration of many data could produce loss of precision and propagation of errors. They have not yet been evaluated in this first version of the accounts.

The use of different sizes of statistical units for data assimilation and reporting produced changes difficult to interpret, as those of ECU values of Artificial areas (cf. Figure 6.9).

Conclusion

In this Chapter, we have integrated the intensity of use and health indexes of each of the three accounts, namely bio-carbon, water, and ecosystem infrastructure, in order to calculate their respective Unit Values, and their subsequent integration in ECU values. Mostly negative changes (degradation) were observed when the analysis is performed by Dominant Land Cover Type. On the other hand, negative and positive changes were found when the calculations are performed by geographic distribution. The negative changes in the north of the watershed from 2000 to 2006 and its recovery from 2006 to 2012 could be related to the decrease in biomass production in 2006. This information could be useful to locate specific areas where decision-making is needed to mitigate degradation.

6.5 References

- Weber, J. L. (2012). Recording ecological debts in the national accounts: possibilities open by the development of ecosystem capital accounts. In *Paper submitted to the ISEE2012 conference, Rio de Janeiro*.
- Weber, J.L. (2014). Ecosystem natural capital accounts: a quick start package, CBD Technical Series No. 77, Secretariat of the Convention on Biological Diversity, 2014, 248 p.

Chapter 7 General Discussion

7.1 Proof-of-concept development

The ENCA protocol, the subject of my thesis, constitutes a bottom-up development of a systems approach to ecosystem capital accounting. The General Introduction chapter has presented the pros and cons of various approaches in the field. I will show below how the ENCA proof-of-concept fulfills the criteria of a strong sustainability methodology and assess the work that remains to be done in order to make ENCA an actionable tool for users in current decision-making. According to the comparative analysis performed in the General Introduction, such an ENCA tool is likely to become a tool of choice in environmental evaluations.

The work I performed was essentially methodological, but I tried to build up my own understanding of how this technical exploratory enterprise can develop cross-disciplinary and policy-making considerations. The elements of a relatively limited conceptual support for this discussion that I present need further reading before going deeper into such considerations.

ENCA results point to a tight and responsible territorial resource management

At the start of the ENCA project there were issues and questions we judged primordial. Such questions were:

- How can human activities be integrated within the limits of natural cycles, functions and services, i.e. how can the resources available in a given territory be managed in a responsible manner?
- How can one understand the recomposition and evolution of territories?
- What new forms of territory could give rise to innovative approaches to reducing emissions, adapting or protecting biological and cultural diversity?
- How can these transformations be integrated into an ecological policy of solidarity and environmental justice?
- How can one combat the increase in territorial, economic, and social inequalities under the pressure of rapid global environmental change (but also social dumping)?
- How can methods be developed to promote the integration of these issues into development strategies and decisions?

Chapter 7 General Discussion

- How can one support local actors in the development and implementation of projects that take into account all the dimensions (ecological, economic, social, cultural, and political) of the transformations in which they find themselves engaged?

The thesis project was aiming answering several of these questions to the extent of reconsidering the way we manage our environment in order to better understand the territorial entities in which novel resource stewardship practices are emerging. This questions the notion of territory (anything from a geographic to a socio-geographic entity; Caron, 2017) and how the Ostrom's concepts of community management of common pool resources (Schlager and Ostrom, 1992) may intersect with the French approach to the territory notion. According to Valette et al (2017), the territory is a framework for development, a space for coordination and regulation in which land use is key to sustainable territorial development.

Moreover, the literature points out the difficulty that arises when having in mind the ensemble territory-landscape-land use (Robert, 2017). The first two are the subject of representations that evolve over time, according to the actors involved and the socio-ecological processes under consideration. With the emergence of the scientific and political challenge of land use, not yet integrated into the monitoring and analysis of landscape changes, ENCA has made it one of its major objectives: understanding territorial and landscape dynamics requires measuring and assessing ecosystem integrity through a series of macro indicators that integrate the land cover artificiality/naturalness, the areas representing high species/habitats diversity and their connectivity (cf. *Chapter 5*).

For example, the ENCA approach points out the fact that the degradation-maintenance of the ecosystem health and the political and economic consideration of the level of intensity of resource use, all resources combined, are essential issues. They can be employed in decision-making to support thinking in terms of territorial capacity, potential, and no net ecosystem degradation (see **Chapter 5, Box 5.1**). Our results show a general erosion / degradation of the ecosystem capital across the Rhône river basin (expressed in ECU over the 2000-2012 period, cf. Chapter 6, Table 6.1) and its specific manifestations according to distinct DLCTs and SELUs (Chapter 6, Figures 6.10 and 6.8 respectively). The results have a double meaning:

- (1) Diminishing ecosystem integrity and their resource potential can but negatively impact their servicing capacity in time;
- (2) The relative reduction of the ECU value on the territory can now be experimented / tested with respect to the corresponding GDP values.

More broadly and with a socio-ecological perspective (Ostrom, 2009), one can take into consideration, for example, how (1) the modelling of the territorial organization can predict its evolution in order to anticipate and plan the sequence of Avoid-Reduce-Compensate (Weber, 2018) or (2) the modelling of inequality and use of resources opens new questions on the collapse and sustainability of societies (Motesharrei et al, 2014; Gylfacon, 2018).

Taken together and on the basis of the ENCA design *per se* (Weber, 2014), my understanding is that the ENCA approach has the intrinsic potential to contribute and bring concrete answers to such conversation or controversy.

More specifically, the purpose of the Rhône watershed ENCA test was to evaluate the relevance and feasibility of ecosystem accounts at a scale which is intermediate between macro approaches aiming at supplementing the national accounts and micro- approaches at the local government or company level.

Regarding the accounting framework itself, and its implementation by spatial units, such as basic hydrological units (UZHYD) and ecosystem natural capital accounting sub-catchments (ENCAT), its implementation is by and large feasible. The balancing set of tables has proved to be useful to cross-check assessments of the various components of the system and to identify data gaps and propose solutions for improvement.

Finally, the ENCA test poses clear issues of collective responsibility to the environment, a broader way to consider the articulation of scales. ENCA has been designed in a multiple scale perspective, including exchanges between socio-ecosystems. This represents the strength of the ENCA methodology, and refinements and consolidation of the protocol are expected to help reveal the nature and the extent of the interconnectedness between processes at work in complex socio-ecosystems.

ENCA, present capacity and future potential

The key point here is the measurement and accounting of the ecosystem intrinsic value that we want to conserve and that has to be conserved. Market prices result from consensus established during transactions. Intrinsic values require as well a consensus on a cost-equivalent, based on a set of criteria to be considered and the way to combine them. While utilitarian values in money express the point of view of the economy, intrinsic values are established from the standpoint of the ecosystem.

The ENCA approach makes it possible to measure the impacts of human practices on ecosystems using ecological assessments based on geographical information. To establish an accounting of the ecological value measured in physical terms, ENCA combines quantitative and qualitative measures (indicators of health, resilience...) of four components: land use, biomass, water, and green/biodiversity infrastructure.

To calculate the ecological value of different ecosystems (their capacity resulting from the state of their building blocks and functioning), their degradation is measured in order to lead economic actors, from governments to companies, to pay the full cost of their use of nature, which implies amortizing the use of ecosystems when there is degradation. It is now an unpaid cost (an externality), not recorded in the price of products incorporating ecosystem services. It should be stressed that the protocol has the inbuilt capacity to evaluate the environmental pressures and impacts embedded in exchanges between a given territorial or political entity and any outside entities.

Stated otherwise, ENCA aims to

- Assess the state of socio-ecosystems in terms of their intrinsic ecological value. ENCA does not say whether an ecosystem is degraded or not, but what its capacity is (its ability to perform its functions over time). It is up to experts to decide whether or not the ecosystem is degraded. On the other hand, a loss of total ecosystem capacity (TEC; see below Figure 7.3) is a deterioration that can be considered in whole or in part as degradation due to human activities. The ENCA accounting is an accrual accounting system whose results are calculated from one year to the next;
- Have economic agents take into account the gains and losses of TEC; in the event of a loss, this leads agents to calculate the amount of repairs (restoration, compensation, etc.). The estimate can be

made by the agents themselves or by the institutions on which they depend on (including the government when it comes to national accounts). The compensations can take a wide variety of forms, at variable costs, and may or may not be effective.

ENCA therefore does not give a target in terms of monetary result but in terms of intrinsic ecological value: no net degradation. It is an invitation to agents to make provisions to amortize their use of nature while leaving open the options that agents have in making their investment decisions. On the other hand, restoration/compensation expenditure is not liberatory in itself, only physical restoration counts. As far as ecosystem improvements are concerned, in the event of a clear and approved improvement, it is a positive contribution to the public good and therefore justifies the allocation of credits in ECU. These credits can be exchanged, probably in the form of a rental. The corresponding market can bring benefits to public and private actors that improve the public good. The incentives for funders are to improve their ecological balance sheet in order to avoid financial penalties.

In brief, ENCA has been designed as a management and control tool enabling to compare and confront through the ECU aggregate estimates of ecological, economical, and monetary values of territorial entities at different geographic and administrative scales.

The integrative and systemic methodological capacity is likely the originality and strength of the ENCA instrument. It can contribute to capture at best a great deal of the interacting processes through which the increasingly dominant influence of human activities are impacting the ecosystem capital. This has been the long lasting challenge in the field, as illustrated by the conceptual model proposed by Vitousek et al (1997) (**Figure 7.1**).

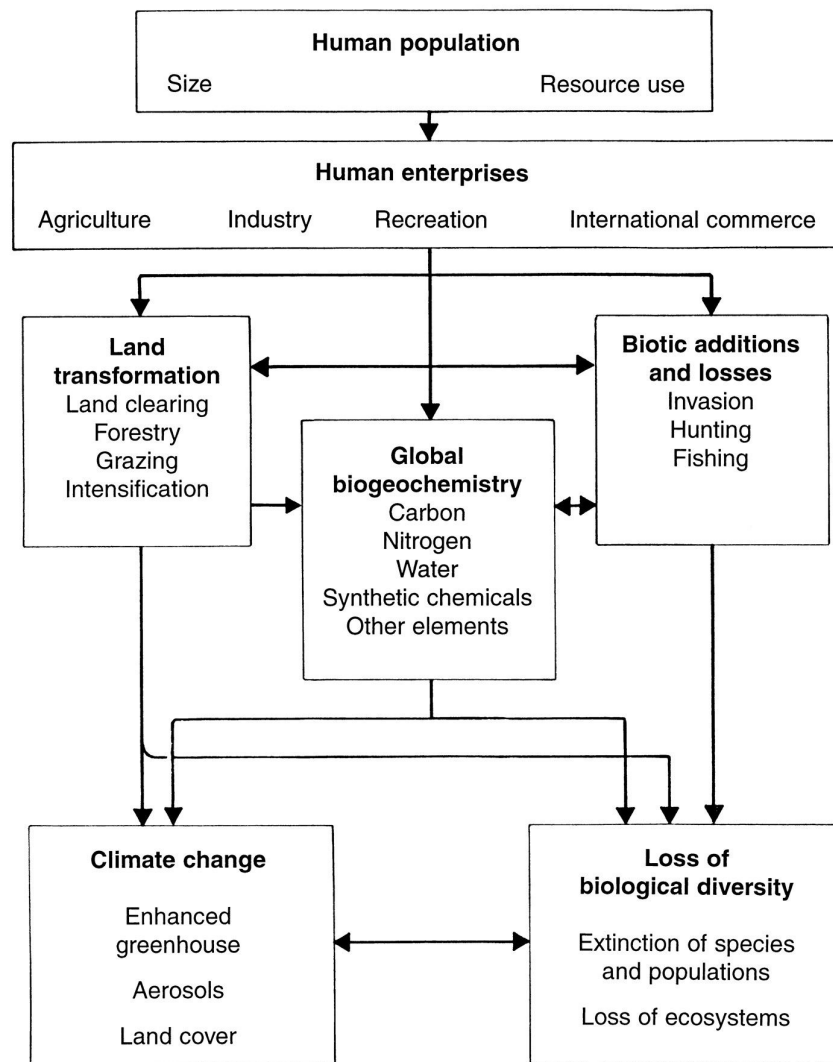


Figure 7.1. Conceptual model (Vitousek et al, 1997) on the growth of human population and the growth in the resource base used by humans in which land transformation strongly interacts with other components of the global environment. Such changes were difficult to aggregate in order to grasp the extent and the rate of cumulative impacts (that could for example sum up in order to estimate trends such as the carrying capacity; see also Ehrlich et al, 2012).

With such a perspective, I have put to test the ENCA potential by

(1) Producing the first exhaustive ENCA tool at the scale of a watershed, the Rhône basin. The tool is at the stage of V.0 proof-of-concept. In brief, ENCA Rhône as it stands today has achieved the following:

1. Collection of geographic information infrastructure (administrative boundaries, watershed delineation, reliefs, rivers, roads);

2. Collection of socio-economic and environmental statistics, land cover data, biocarbon account data, water ecosystem account data, and green infrastructure account data. Data quality control and cleaning;
3. Creation of accounting tables through extracting information from the spatial data base;
4. Create GIS projections to generate account reporting and integrate them into higher order indexes;
5. Establish a first accounting evaluation of the ecosystem capital of the Rhône river watershed in order to identify the magnitude of changes in the ecological state values (tables and graphs) and identify critical areas (that should make the object of more specific local analysis);
6. Generate an integrated ecosystem capability outlook and its expression in a final unit, the ECU. The capability approach implies producing a whole range of intermediate set of indexes that inform on the accessibility (e.g. NATURILIS), intensity of use, and the integrity of various resource systems that constitute the ecosystem capital.

Alltogether, I managed to work out Rhône river watershed proof-of-concept that raises a large body of questions and deserves being questioned in all its components. Before making more elaborated considerations than those that follow, this proof-of-concept V.0 requires a global evaluation through the generation of V.1 version. The summary of these results is illustrated in **Figure 7.2**.

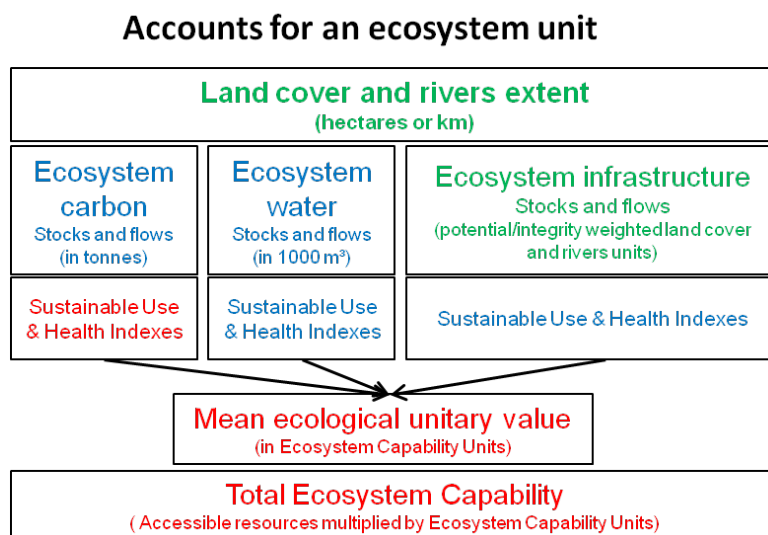


Figure 7.2. ENCA diagram indicating through a color code the articulation of relatively robust (green), satisfactory (blue), and fragile (red) components.

In this report, the presented results represent the *ad litteram* transposition of the ENCA protocol. A series of inconsistencies need to be highlighted, of which best examples are the production of the Biodiversity Index and the effects on SELUs of the artificialized areas. Next step would consist in working on each of these issues in order to generate alternative solutions.

(2) Sketching how ENCA generates information to integrate the major objectives of various institutional and policy agendas (such as UN-SDGs, EU and OECD, IPCC etc; **Figure 7.3**) at a time when such agendas remain largely sectoral or poorly interconnected.

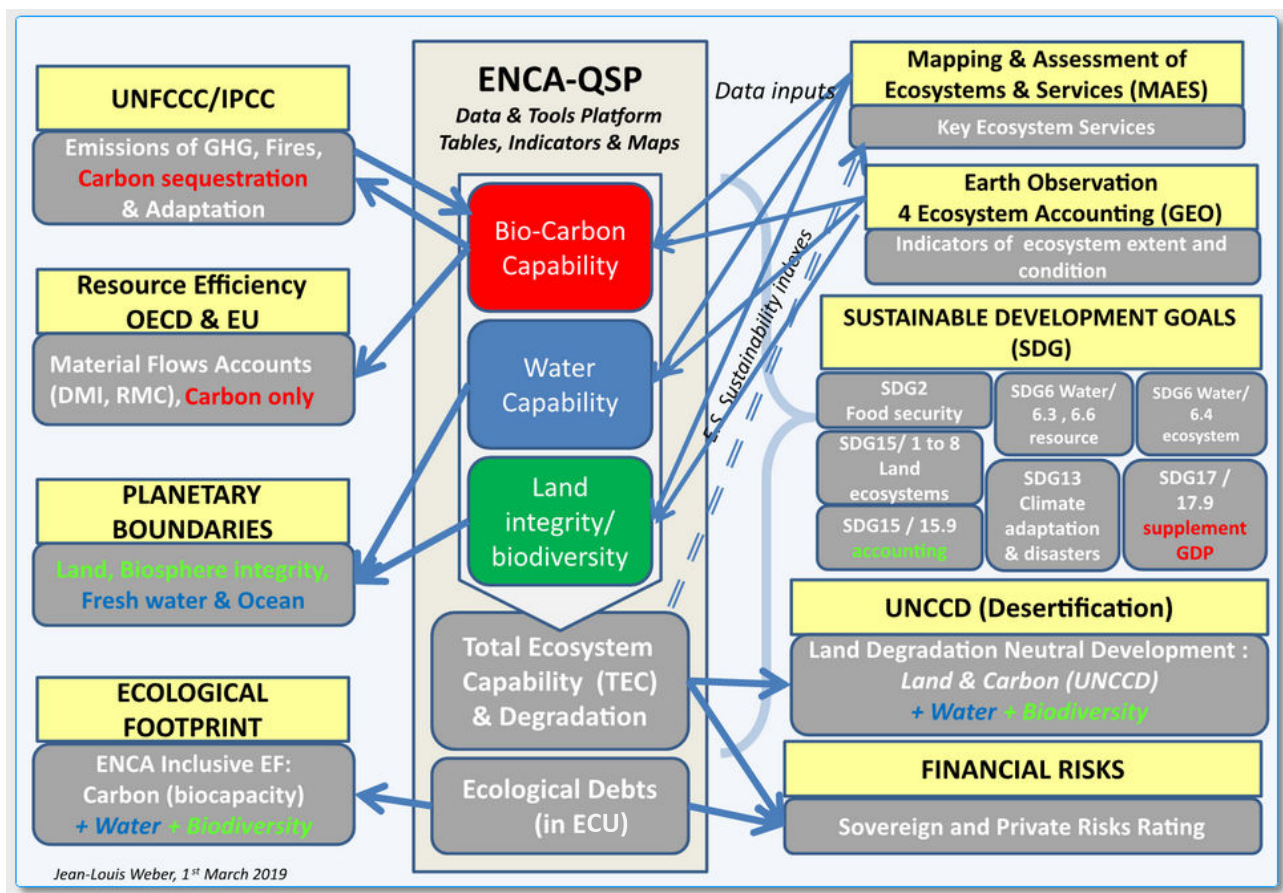


Figure 7.3. ENCA, an integrator of environmental indicator systems and ecological transition priorities, namely land / soil, water, and biomass. The arrows show how the ENCA components connect with various environmental and policy agendas (Courtesy of J.L. Weber).

Taken together, the above considerations indicate that ENCA approach has an open potential to responding to the needs and expectations of both public and private stakeholders. To validate the full potential of the ENCA tool in real life conditions, it is important to optimize the present proof-

of-concept, and to deploy ENCA at national (such as France-ENCA) and planetary (GlobENCA) scales. The two latter aspects are presented in Chapter 8 (*Section 8.2*).

ENCA proof-of-concept limitations and how to circumvent them

The results of the distinct ENCA Rhône account components point to the following series of limitations, obstacles, or handicaps. They are either account-specific or more general. **Table 7.1** summarizes the identified problems in each category of accounts.

Table 7.1. Obstacles in the production of ENCA accounts

Obstacle designation	Land use	Bio-carbon accounts	Water account	Eco-infrastructures and synthetic indicators	Summary scores
Resources in space					
Grouping of Land cover classes	X				1
Grouping of stocks and flow categories	X				1
Designing flow classes vs vegetation types	X				1
Spatial unit designation	X		X	(X) ²	3
Data sources					
Satellite data limitations		X	X	X	3
Data sources accessibility		X	X	X(X)	3
Lack of time series		X	X	X(X)	4
Incomplete (gaps) and quality of data	X	X	X ¹	X(X)	5
Multiplicity of data providers		X			1
Heterogeneity of data		X	X	X	3
Scale and interoperability		X	X	X	3
Analytical instruments					
Heterogeneity of geoprocessing software and tools	X	X	X	X	4
Geoprocessing errors		X			1
Indicators and indexes. Scoring criteria and statistical models			X	X	2
Assumption and normalization		X	X	X	3
Score (per account category)	6	10 ³	10 ³	13	

Notes

- (1) water accounts - data for accounting proved to be fairly abundant and of good quality, including quality geo-spatialized data, but problems with water use, management, and distribution.
- (2) When in brackets, the impact of that obstacle is indirect.
- (3) The amount and relevance of data analysed for the two accounts allow to score them as roughly equivalent.

This across account analysis shows that the main obstacles to producing the ENCA Rhône proof-of-concept are:

1. Data sources
2. Analytical instruments
3. Resources in space.

On the basis of this analysis a general recommendation stands out: the urgent need of implementing coherent data policies. In the absence of such policies, the implementation of any ecological assessments is likely to remain a hazardous undertaking.

1. Conceptual and more general methodological obstacles

ENCA has his roots in the UN-System of Environmental and Economic Accounting (SEEA) framework and in particular in its “experimental ecosystem accounting” (SEEA-EEA) guidelines. The methodological inconsistencies, the absence of integration of various accounting domains and the relatively weak capacity to support implementations by SEEA-EEA (see *Suppl. Table 1.1, Chapter 1*) have been taken into consideration in developing the ENCA approach. Has the ENCA Rhône work been able to sufficiently address these genuine and other internal limitations? Here I attempt a brief critical analysis.

The terms of reference of ecosystem capital accounting is to bring together datasets of different nature, more or less heterogeneous. Putting things together is feasible because all data reflect the same system, seen from various angles: natural assets, natural flows, artificial flows, withdrawals of resources... They exist in due proportion to each other and inconsistency can be detected by maps and accounts, and carefully reduced when proceeding to what national accountants name “arbitration”. They are therefore iterations in compiling ecosystem capital accounts: collecting best

available data, formatting them with common rule for making comparisons possible (data assimilation), and adjust them when needed to match the broad equilibrium of flows and stocks.

Furthermore, even if the technical dimension is central to the ENCA accounting exercise (data reprocessing, temporal consistency of intermediate results, parameters and calculation methods, etc.), ENCA allows - by its exhaustiveness - to directly refer to and include the geographical and ecological sciences when linking and putting in perspective its core components, namely

- ways to represent, observe and quantify the ecological status of rivers,
- changes in land use associated with changes in landscape properties, and
- dealing with the bias of geovisualization in terms of uncertainties in the representation of data (see for ex., Arnaud and Davoine, 2019)

2. Obstacles related to data

The use of accessible data has been the rule in working out the proof-of-concept tool. The input data and the modalities of information production (spatial data entry, consideration of metadata, etc.) constitute the main basis of the methodological protocol:

- data sets (vector maps, raster, sampling data, etc.);
- very diverse data (satellite, VGI, surveys, etc.), whose heterogeneity and production contexts must be taken into account.

This has to be understood as “accessible under the conditions of the project”. Indeed, other existing datasets could have been used, and should definitely be used for future implementation, but their access is presently restricted either because they are sold on a commercial basis or because of data policies limiting public dissemination. On this last point, huge progress has been done in the last decades but some public organisations keep a privative attitude regarding the data that they collect for their mission or research and give access to data only under the form of printed or .pdf maps which cannot be used for GIS calculations. Regarding data hold by private companies, it is likely that their contribution to the ENCA project could be beneficial; it requires however efforts which were beyond the time budget of the whole project. This is important matter for the future, considering in particular water distribution.

An additional point relates to the time resolution of the data. Accounts are expected to be produced on an annual basis to meet the calendar of economic accounts used for decision making.

This is not an obvious matter when it comes to the relatively general obstacle of data production and access. However, there are developments under way that suggest that annual ecosystem capital reporting is possible. The results of land use changes (spatial analysis, nomenclature comparison systems, matrix analysis indices, etc.) should benefit from the context (Land Use, Land Use Change and Forestry) within the framework of the United Nations Convention on Climate Change and the Kyoto Protocol, for which land use changes and associated carbon flows are accounted for annually (Robert, 2017).

Based on the above considerations and on the significance of the results of the proof-of-concept as reported here, a major contribution of my work consisted in analyzing in detail each category of ENCA core accounts. On those grounds, I could make some recommendations on the most appropriate periodicity of data production for a complete set of variables. It seems likely that relevant changes in ecosystem capital parameters occur with different time periodicity. For example, changes in carbon soil content operate with a different timing as compared with the agricultural production or the ecological state of rivers. The production of the corresponding data sets should take that in consideration and should make the object of public policy data protocols.

In addition, I have identified a significant set of barriers and data inconsistencies (cf. **Table 7.1**). This compilation of available data and their metadata allowed and – whenever I was not able to perform control tests, should allow capturing in a first approximation the differences in relevance of the various sources by evaluating

- the causes of the series biases (for example, the first three years of NPP time series from NASA, and the annual series estimation of the Biodiversity Index);
- uncertainties about the quantification of certain stocks or flows (for example, coefficient for the conversion of animal biomass to carbon, soil carbon, biomass, etc);
- the causes of uncertainties in the source data and redefining a methodological framework to estimate more relevant rates of change. This depends mostly on the precision of collected and used data;
- whether the interpretation generated errors, false negatives and false positives needs to be evaluated, particullary, the Biodiversity index and the ECU value in Artificial areas.

On these lines, the literature drew my attention on two recommendations (Robert, 2017):

- To best meet the needs of the accounting inventory, relevance is preferred to accuracy, and scale dependence effects should be extrapolated with caution;
- Test the hypothesis of false changes through field tests and surveys to better understand (un)avoidable errors.

Finally, ENCA allows formulating an integrated perspective of the data used within a territory based on heterogeneous data sets and reference frames, and thus to weight the estimates obtained with regard to their uncertainty, their fields of validity, etc. In this way, ENCA contributes to putting into perspective the methods and concepts of the accounting inventory.

More generally, one key issue is how to make data intensive analysis (integration, organization, and visualization) of multi-scale and multi-sector information tractable to testing hypothesis in environmental and socio-ecological research? Assimilation of large and heterogenous data should improve modelling approaches that inform spatially explicit frameworks. One example concerns life cycle assessment modelling to simulate the value of biodiversity and ecosystem services (Rugani et al, 2016). In their work, MIMES (Multi-scale Integrated Model of Ecosystem Services) was adapted to solve problems related to collecting, elaborating, managing, and running large sets of data while incorporating additional data sets and meta-data layers (**Figure 7.4**).

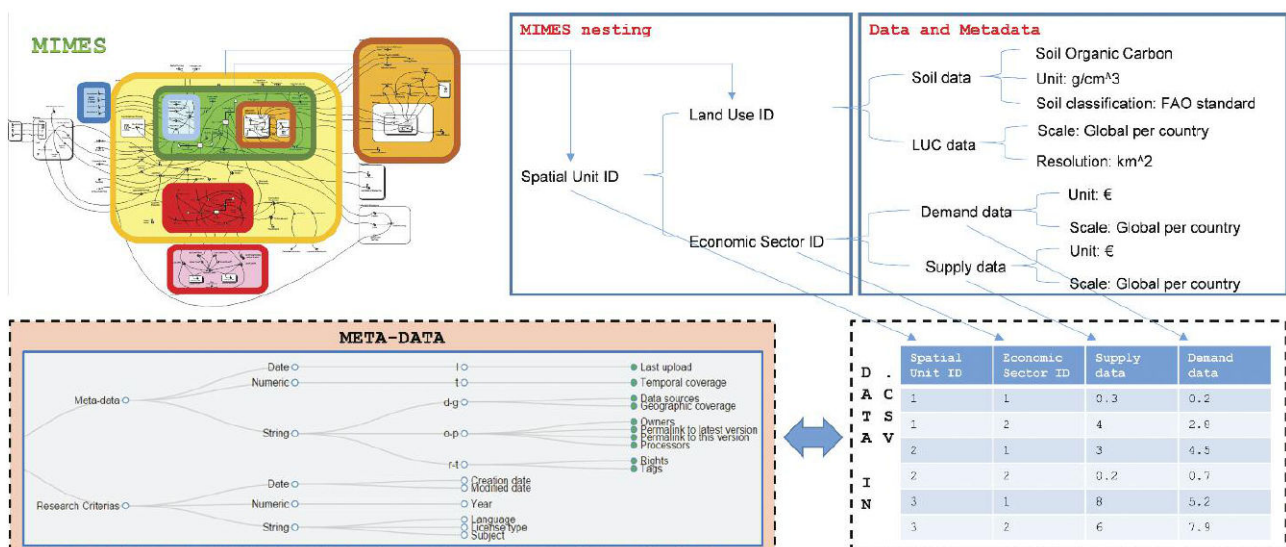


Figure 7.4. Generating a unique modelling framework (Rugani et al 2016). Illustration of how the existing meta-data on the association of species and habitat types to ecosystems in the EU is stored in a modular system and subsequently upgraded with additional information in order to be harmonized in one unique modelling framework.

To conclude, I consider that my work is open to further discussion because the production of the proof-of-concept has been time consuming and I could not develop systematic alternative approaches and comparative tests to challenge the genuine ENCA system. All in all, it looks to me that ENCA is a pragmatic approach to managing the uncertainties of input data, and is built on a logic of continuous improvement through a succession of precision additions that ultimately provide greater robustness to justify relevance gains.

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Chapter 8. Perspectives

What follows is a summary of conversations, internal meetings, seminars, and discussions I had with members of the pedagogic team and members of the PhD follow up committee.

8.1 Transborder ENCARHÔNE

The Rhône watershed is a continuous ecological entity. The proof-of-concept V.0 is restricted to the French part of the basin. Contacts have been initiated in 2016-2017 with the OFEV/FOEN, the Swiss Federal Office for the Environment, with the support and mediation of EPFL, LASIG and Green Economy laboratories.

Table 8.1 summarizes the contacts with different sectoral OFEV services. So far, we were unable to complete the ENCA accounts on the entire Rhône watershed because the access to sufficiently elaborated data, in particular river/water and agricultural biomass, was not possible. This remains to be accomplished in a second stage through a memorandum of understanding with the OFEV. The present proof-of-concept might serve as a tutorial in attempting to circumvent this obstacle.

Table 8.1. Data requests made to OFEV/ FOEN.

1	Forest maps (beyond CLC), wood and alignments (French <i>alignements</i>).
2	Standing wood volumes, table and fine mapping (hardwood, coniferous, mixed / forest and alignment (French <i>alignements</i>)).
3	Harvesting of hardwood (hardwood, coniferous / roundwood (work and crushing), firewood).
4	Agricultural statistics, if possible by municipality.
5	Agricultural land plot with crop type indications and irrigation.
6	Indicators and ecological fragmentation / integrity maps.
7	Ecological indicators and status cards broad sense.
8	Ecosystem accounts as such (for example, the idea of applying the planetary boundaries approach to Switzerland).

As a matter of fact, FOEN has been engaged since 2015 with transposing at the country level the environmental limits and Swiss footprints based on the Planetary Boundaries approach (Dao et al, 2015). This is one reason to co-organize with the Swiss partners the 2020 conference on “Environmental impact assessment methodologies – societal and political demand, and state-of-the-

art debate” in order to confront the two approaches on the same ecological entity, the Rhône watershed (see **Suppl. File 1.1** in the **Chapter 1**, General Introduction).

Tool optimization

This is envisaged in two steps.

1. Publish the results while testing the proof-of-concept with a large body of potential users, namely

- insurance and financial bodies, such as Swiss Re, Caisse de Dépôt et Consignation (CDC, a territory support financial institution),
- risk evaluation and rating agencies (IRGC Lausanne, Beyond Ratings,)
- governmental agencies, private, and private-public organizations, such as VITO Belgium, France Stratégie, Plan Ressources pour la France, AFD – the French Developmental Agency, AFB/ONEMA (the biodiversity agency), regional providers of data and census activities (Agence de l’Eau, DREAL, DRAAF, ADEME); Natural Capital Coalition : UNEP, CNR, Suez-Engie, ...)
- research programs on the subject (MNHN, IRSTEA, ZABR, ...)
- NGOs (WWF, FNE, AFB, ...).

2. Automatization of the current protocol. The automatization is an important element in demonstrating the ENCA technical capabilities and interest to the users. The automatization of calculations for ecosystem accounts (statistical and geographical data) is necessary to simplify the treatment chains by:

- setting-up a spatial database and geo-server;
- developing automated Python workflow to transform input data into grid and integrate gridded data into the spatial database for accounting purposes.
- integrating them into a single environment (e.g. jupyter notebook) to facilitate their exploitation. The process consists of integrating the modules of the various geomatics and spatial analysis software packages (e.g. GDAL, QGIS SAGA-GIS).

8.2 France ENCA

Through its systemic dimension, ENCA is able to inform governance on the coherence of public policies, the nature and timing of risks, and geopolitical issues related to national security and sovereignty. Tools of this kind are supposed to enable France to position itself in the changing environment of a world order in which renewable ecosystem resources will play an increasingly strategic role. Notably, France has an important ecosystem capital base, in particular for the triad land-water-biomass.

Is France involved in such an approach?

Data creation projects are numerous (see for example the seminar of the Agence française de la Biodiversité, AFB 2017) and the institutional context is also undergoing important changes. In France, the role of the national public authorities has been major in the establishment of standards; however today the projects are split between decentralized powers, European and international, or between public and private organizations (with notably the role of multinationals, such as Google).

In 2015, the National Assembly adopted the billtaking into account new wealth indicators in the definition of public policies (No. 2285; <http://www.assemblee-nationale.fr/14/ta/ta0469.asp>).

In 2015, the Senate decided to produce the "Resource Plan for France" (to be updated every 5 years). "Today, no country in Europe has a forward-looking clear vision of its needs and the availability of its own resources," argued Senator Chantal Jouanno. In 2017, the draft National Biomass Mobilization Strategy (SNMB) plan was finalized. The plan was built not by sector of activity but by resource categories (mineral, non-energy, biomass, and soil). It identified knowledge gaps regarding resource flows, and estimated future needs. It was seeking to take into account the interactions between resources through coherent and articulated resource policies (renewables in particular) and pointed to the urgent need for diagnostic tools to move from observation to integrated management (https://fr.wikipedia.org/wiki/Plan_of_resource_programming).

In 2018, the new CNIG / IGN geo-platform program for sovereign geographical information at the regional level was launched, aiming at allowing a description of the territory for all strategic

resources (physical, digital, institutional). As a design of new services and added value, the proposed tools are expected to promote interoperability around major equipment projects. The main lock is the fact that primary data providers are not (yet) in this process.

ENCA can inform on matters of national security

Environmental assessments are currently quite accurate when referring to variables such as material flows, atmospheric emissions, energy and water stocks and flows (in physical and monetary terms). Ecosystem integrity and biodiversity remain more difficult to measure (Mazza et al, 2013), but the rapid development of satellite and *in situ* monitoring and data processing systems is changing the situation by allowing changes (degradation or improvement) in renewable resources to be monitored in near-real time. This is why the control of digital resources and their regular use in the assessment of physical resources are necessary to assess risks, guide investments, and manage natural patrimony and territories (UNEP 2012; FAO, 2011). The issues are therefore a matter of energy and food security, of national sovereignty, particularly in view of the conceptual and operational advances at work in the private sector: integrated environmental diagnostic systems are proposed by platforms managed by operators specialized in the production of massive data, such as Google Earth Engine, but also, more recently, the European network Earth Intelligence and Planetary health intelligence with 5 platforms for the exploitation of Copernicus data (see also ***Supplement. File 1.1, General Introduction***).

In conclusion, the interest of the ENCA Rhône in this context is to make understand, based on the results I generated, the stringent necessity of top-down data policy design because producing systematic and exhaustive ecosystem capital accounting is much about national security.

8.3 Global ENCA

A global ENCA tool would accelerate the harmonization of a series of instruments, norms, and standards that are required to switch gears in developmental policies, namely the extension of the polluter pays principle to ecosystem degradation, to green finance, to sovereign and private financial risks rating and integration of Environmental, to Social and Governances (ESG) risks

factors (OECD, 2017) in investment decision, to the estimation of commodities full price by internalising ecosystem depreciation, etc.

GlobENCA would implement a first generation of ecosystem natural capital accounts at the global scale. Best available data and statistics can be assimilated into a 1 km × 1 km grid allowing the production of thematic maps of the various variables, bio-carbon, water, and functional infrastructure accounts in particular. Such a tool would match the challenges put forward by the global market and would supplement the IPCC carbon budgets with variables on water, ecosystem functions and biodiversity. GlobENCA would help framing global policies and boost their implementation with relevant, comparable, and verifiable and updated data.

GlobENCA should benefit from global scale reports and associated data sources on biomes-anthromes (Ellis et al, 2010), land use and soil, agriculture, water, consumption patterns, energy, etc. evaluations (see for example Good and Beatty, 2011; Ramankuty et al, 2008; Liu and Yang, 2010; Hookstra and Mekonen, 2011). The global biomass-water-land issues and corresponding additional references are synthesized in *Supplementary Table 8.1*.

In support to these arguments showing the extent of and why such primary resources are under stress, exhaustive and periodic evaluations are needed. GlobENCA can be tailored to integrate these dimensions and have his word in the frame of the nowadays geopolitics of resource scarcity (Acunzo et al, 2018). A GlobENCA is finally important today in positioning the ENCA approach among the main providers of ecosystem capital information.

Supplementary Table 8.1 A general landscape intersecting and integrating the main trends of the three primary and interdependent ENCA resource component - land, water, and biomass. Soil, nutrients, and water participate together in global NPP and crop productivity. Improving crop use efficiency of these resources and their conversion into biomass is essentially a research issue. The most urgent actually is to provide a responsible and integrated management of land, water, and biomass. The triad is a buffering system at hand in the face of how humans are changing the geo-physical conditions of the planet (Steffen et al, 2015) with direct impacts on water, nitrogen, carbon etc. cycles in the biosphere (OECD-FAO, 2012; Foley et al, 2005 and 2011). This is particularly critical in a context in which 25% of agricultural land is highly degraded, with irrigation capacity reaching its upper limit (Tanentzap et al, 2015) while facing water scarcity (Mueller et al, 2012; Fenichel et al, 2015).

<i>Biomass (food supply, etc), and footprints</i>	<i>Examples of sectoral solutions</i>
<p>174 crops traced by FAO (Foley et al, 2011).</p> <p>Prominent changes in yield patterns with reduction mainly of fodder crops and increase in oil crops (Foley et al, 2011)</p> <p>Crop allocation disparities (cash vs staple crops): 40% cropland to food in Europe and North America versus 80% in Africa and Asia (Foley et al, 2011)</p> <p>Agriculture expansion mainly in the tropics, with 80% of new croplands replacing forests: contributed by 22% to increase in production but little to global food supply (Foley et al, 2011).</p> <p>Further expanding agricultural land will increase agriculture C footprint. Potential new land has low productivity (forests and savannah in S. America and Sub-Saharan Africa) (FAO, 2011).</p> <p>Production gains through agriculture intensification (increased energy use, water degradation, global pollution and GHG emissions, disruption of N and P cycles (Foley 2011). Strong feedbacks between further expanding of industrial agriculture and critical resource scarcities (FAO, 2011).</p> <p>Yield ceilings and gaps and production loss estimates in major crops due to soil degradation average $0.3\%yr^{-1}$ at the global level (den Biggelaar et al, 2003; Wiebe, 2003).</p> <p>10% of global energy use is for smallholds and depends on traditional biomass.</p> <p>Competition food-bioenergy estimates: liquid biofuels feedstocks to double by 2030 (>3% of cultivated land).</p> <p>Coping with climate deregulation results in changes in root and leaf</p>	<p>Shifting diets and changing food systems (Foley et al, 2011; De Schutter, 2017)</p> <p>Increase cropping efficiency, crop diversity and closing yield gaps on underperforming soils (Foley et al, 2011).</p> <p>Diverse agro-ecosystems in which monitoring of total productivity per acre is replacing single crop yield per acre.</p>

<p>biomass. Such changes are variable among species, including C3 and C4 species, and affect phase transitions, and reproductive development with incidence on grain yields (Gray and Brady, 2016).</p>	
<p><i>Water use and footprints</i></p>	<p><i>Examples of sectoral solutions</i></p>
<p>Irrigation was a major component of the Green Revolution allowing to triple grain productivity between 1950 and 2010 (Brown, 2011c). Since 1950 water consumption grew twice as fast as the global population (Atlas, 2007; Brown, 2011b).</p> <p>70% increase in irrigated cropland area (Foley et al, 2005).</p> <p>Agriculture consumes 50-70% of total global ‘blue water’ withdrawals (FAO, 2011; Foley et al, 2011) and water use is heavily associated with industrialized grain-fed livestock systems (Godfray et al, 2010).</p> <p>Irrigated areas concentrated in South and East Asia and USA (Mueller et al, 2012) with rapid increase in SE Asia and South America (FAO, 2011). Irrigation is presently used on about 20% of cropland (with differences as high as 10% of Africa’s farmland, 26% in India and 44% in China) and generates 34% of agricultural production and 50% of food production (FAO, 2011; Brown, 2011c). Water efficiency is low (Foley et al, 2011).</p> <p>Water shortages affect 20 countries containing half the world population, of which the three big grain-producing countries - China, India and the US. On the short run, water shortages affect food security in the Middle East (Brown, 2011b and c). (<u>Note 1</u>)</p> <p>Shortages are amplified in many countries with low levels of fresh water through trade of water “embedded” in agricultural export (Chapagain and Hoekstra,2008).</p>	<p>Coordinate and prioritize water demand: expected rise for agriculture by over 30% by 2030 and could double by 2050 owing to pressures from industry, domestic use and the need to maintain environmental flows (FAO, 2011).</p> <p>Reduce unsustainable water withdrawals and phase out water pollution from chemicals.</p>
<p><i>Land-soil-nutrient and footprints</i></p>	<p><i>Examples of sectoral solutions</i></p>
<p>38-40% of land surface as cropland and pastures with large increase in energy, water, and fertilizer consumption. Fertilizer excess in China, India, USA, western Europe (Foley et al, 2011; FAO, 2011 and 2016; <u>Note 2</u>).</p> <p>Land degradation disrupts biological cycles and lessens ecosystem services. Agriculture is confronted with severe topsoil erosion. While natural regeneration rate dynamics can vary from 50 to 1,000 years, it is accepted that one-third of the world’s arable land has suffered erosion during the last few decades (Pimentel et al, 1995; FAO 2011), that 40-50% of global croplands experience soil quality constraints (Foley 2005; FAO, 2011), and</p>	<p>Good practice agriculture: organic matter, soil structure (C sequestration) and biodiversity conservation, with beneficial effects on water-holding capacity, nutrient balance-availability-</p>

<p>that desertification is globally progressing (gov.uk/Forsight).</p> <p>Despite difficulties in properly assessing the extent, rates, and impacts of soil erosion, such degradations are estimated at 10-17% of global GDP (ELD initiative, 2015).</p> <p>Soil degradation mapping in high crop yielding regions (Banwart, 2011; Foley et al, 2011) shows effects from overgrazing (34%), deforestation (30%) and removal of the natural vegetation, agricultural activities (28%) and improper management of agricultural land, overexploitation of the vegetative cover for domestic use (sealing), and industrial activities leading to chemical pollution (GLASOD, 1990; Scherr, 1999). Intensive cereal farming accounts for humus loss at 3% year rate (Banwart, 2011).</p> <p>Large scale acquisition of cropland (land grabbing), with main targets Sub-Saharan Africa, SE Asia, and Latin America (FAO, 2011).</p>	<p>efficiency, ecosystem services capacity (Foley et al, 2005, 2011)</p> <p>Switching to conservation agriculture and agro-ecology through management of crop residues, permanent soil cover, systematic crop rotation.</p> <p>Achieving land-degradation neutral societies</p>
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Integrating socio-economic and policy levers (balanced rural development: land use and water management)

Climate change puts productivity under stress requiring changes in water availability, soil conditions, and requirements to close yield gaps (Foley et al, 2011).

Economically speaking, water and soil resources have little likelihood of being both outsourced and substitutable by other forms of capital.

Adaptive water management (community, public good).

Gains in water efficiency / productivity are slow; make sustainable use of water to avoid future drops in crop productivity through smart irrigation and precision agriculture with solutions and innovations available to smallholder farmers (Morison et al, 2008).

Increase investments in agriculture water infrastructures and management, and provide access to water and water rights (FAO, 2011).

Policies on water quality, aquatic ecosystems, resource-efficiency, equitable management (UNEP, 2007).

Improving water conservation and adapting crops to water availability and climate conditions has a direct impact on better soil conservation (FAO, 2011).

While there are instruments to regulate air and water quality (e.g. EU water directive, 2013/39/EU), soil has so far not been subject to a coherent set of rules in the EU (http://ec.europa.eu/environment/soil/index_en.htm).

Land use and soil management

Unwise land use decisions, management practices or financial incentives may have greater impact on land degradation than extreme natural events or climate change (Boardman, 2006; den Biggelaar et al, 2003; Richter and Markewitz, 2001; FAO, 2011).

Coherent and strong policies determining the optimal allocation of land to different uses (FAO, 2011).

Improved governance of land and water resources by integration of policies combined with investments in food security and poverty alleviation (UNEP, 2007).

Social and economic triggers

Management changes through social triggers and practices that improve trade-offs between production and environmental factors (Foley et al, 2011).

Investment in ecosystem resilience (landscape connectivity, wetlands restoration etc) and strengthening of rural communities (De Vivo, 2016)

Networking of practitioners sharing knowledge and tools (platforming) (Thompson and Schauer, 2016)

Review of mandates and activities of global and regional organizations for land and water, and basin-wide management institutions in order to promote integration (UNEP, 2007)

Note 1. The state of “water stress”: the corresponding indicator measuring the availability of minimum 1700 m³ of fresh water per capita per year is not attained by 40% of the world population. The deficit is expected to reach more than 60% by 2025, the amount of water losses being estimated at 30% between now and 2025 (Atlas, 2007).

Note 2. Fertilizer use: in 2002 China used 30% of N- and P-fertilizers produced worldwide, although its arable land accounts only for 10% of the world total (i.e. 75% above average use <http://faostat.fao.org/>), and N-use efficiency is low (Zang, 2007; Good and Beatty, 2011).

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This PhD work is an attempt to coherently link the increasingly environmental concerns of the society with the academic and institutional research efforts that analyse and evaluate the impacts of human activities on the natural capital.

For the society at large, the dominant issues are those that question what needs to be done to counteract the effects of climate change and to halt the erosion or even the possible collapse of biodiversity. Or how to implement the energy transitions.

These questions are important ones, but are they relevant enough to work out acceptable solutions that halt the degradation of nature, the overconsumption of physical resources, the unbridled growth etc., all driven by a "low cost" economy? These questions are most likely more relevant in terms of how to entirely rethink the development system of our societies. The corresponding policies ought to concentrate first on conceiving ways to evaluate the ecological impact costs of human activities, so as to learn to use natural resources well rather than exploit them, to put them at the service of collective needs rather than individual profits, to renew them rather than consume them.

Such societal readjustments require a political approach based on scientific evidence-based decision-making. It turns out that the environmental assessment research is a fairly recent research area that is constantly evolving and developing new approaches and methodologies. To the point of making it difficult to clearly grasp their real objectives and how these methodologies and tools serve the public rather than the particular interests. It must be recognized that the ecosystem capital problematic is probably one of the most complex intellectual work one can deal with: how to amortize an ecological capital currently consumed that is rather poorly defined as negative externalities? Most of the time the "polluter pays principle" is far from integrating the inseparable bundles of drivers, impacts, and responses at work in real life situations.

My thesis work aims to tackle such problems by putting to test the methodology known as "ecological natural capital accounting". This is one approach among many others. Concerning accounting, all-and-every-one practices it individually or collectively, and states have been using national accounts since a long while. But when it comes to accounting (on) nature, the task is rapidly becoming a very ambitious enterprise, even in the digital era: it involves the working together of diverse disciplines along with all sectors of activity and governance.

Attempting to do it at present sounds like a double commitment, requirement, challenge, and responsibility:

- (1) Nowadays knowledge and techniques allowed to walk on the Moon. They should also allow to be accurate while walking on the blue planet. Actually, "the blue planet is a green world";
- (2) The society is increasingly aware that the ecosystems are in fact the life, health, and well-being support on which it depends on.

I undertook the challenge and managed to develop the research protocol to the stage of a proof-of-concept tool, enabling me to offer the first rather elaborated experimental ecosystem capital accounting tool for a broad range of users and stakeholders.