

1 **Operationalizing Ecosystem Services for the Mitigation of Soil Threats: A**

2 **Proposed Framework**

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28

29 **Abstract**

30 Despite numerous research efforts over the last decades, integrating the concept of ecosystem
31 services into land management decision-making continues to pose considerable challenges.
32 Researchers have developed many different frameworks to operationalize the concept, but
33 these are often specific to a certain issue and each have their own definitions and
34 understandings of particular terms. Based on a comprehensive review of the current scientific
35 debate, the EU FP7 project RECARE proposes an adapted framework for soil-related
36 ecosystem services that is suited for practical application in the prevention and remediation of
37 soil degradation across Europe. We have adapted existing frameworks by integrating
38 components from soil science while attempting to introduce a consistent terminology that is
39 understandable to a variety of stakeholders. RECARE aims to assess how soil threats and
40 prevention and remediation measures affect ecosystem services. Changes in the natural
41 capital's properties influence soil processes, which support the provision of ecosystem
42 services. The benefits produced by these ecosystem services are explicitly or implicitly valued
43 by individuals and society. This can influence decision- and policymaking at different scales,
44 potentially leading to a societal response, such as improved land management. The proposed
45 ecosystem services framework will be applied by the RECARE project in a transdisciplinary

46 process. It will assist in singling out the most beneficial land management measures and in
47 identifying trade-offs and win-win situations resulting from and impacted by European
48 policies. The framework thus reflects the specific contributions soils make to ecosystem
49 services and helps reveal changes in ecosystem services caused by soil management and
50 policies impacting on soil. At the same time, the framework is simple and robust enough for
51 practical application in assessing soil threats and their management with stakeholders at
52 various levels.

53

54 **Key words:** ecosystem services, soil functions, soil threats, land management, decision
55 support, Europe

56

57 **Highlights**

- 58 • Integrating ecosystem services into land management decision-making is a challenge.
- 59 • An adapted framework for soil-related ecosystem services is needed; we present one.
- 60 • It helps identify changes caused by soil management and policies impacting on soil.
- 61 • It will be used to single out the most beneficial land management measures.
- 62 • Consistent terminology and clarity enable practical application with stakeholders.

63

64 **1. Introduction**

65 The mitigation of soil threats – such as erosion, compaction, salinization, sealing,
66 contamination, or the loss of organic matter, to name just a few – is an increasingly
67 challenging task for the global community, especially in light of population growth and

68 climate change. Productivity goals related to immediate human needs often negatively affect
69 long-term environmental sustainability (Foley et al., 2011). The concept of ecosystem
70 services describes the benefits people obtain from ecosystems (MEA, 2005) and is suitable to
71 illustrate the dependence of human well-being on ecosystems. Considering ecosystem
72 services is thus crucial when improving agricultural production systems in order to reduce
73 yield gaps (Bennett et al., 2010; Bommarco et al., 2013). In addition, soils, being part of the
74 natural capital, provide or contribute to a multitude of ecosystem services that range far
75 beyond agricultural production. Without the ecosystem services provided by soils, for
76 example, we would have no clean drinking water, nor adequate protection from floods.
77 Nonetheless, the various values of soils are often underestimated (Robinson et al., 2014) and
78 remain largely unrecognized.

79 Given the importance of soils, their protection has enormous significance for human well-
80 being and our social and economic development. To date, however, land management
81 planning and the implementation of practices to mitigate soil threats do not take sufficient
82 account of ecosystem services provided by soils (MEA, 2005; Schulte et al. 2014, FAO and
83 ITPS, 2015). Efforts to use soil sustainably and preserve its ecosystem services are at the core
84 of the EU research project RECARE (Preventing and Remediating Degradation of Soils in
85 Europe through Land Care, 2013–2018, www.recare-project.eu). To this end, RECARE aims
86 to measure how soil ecosystem services are affected by degradation and conservation.
87 RECARE is engaging with stakeholders in a transdisciplinary process to develop and select
88 appropriate methods to measure, evaluate, communicate and negotiate the services we obtain
89 from soils, with the ultimate aim of improving land management. This research process
90 requires a sound understanding of the ecosystem services concept and the current scientific
91 debate on the assessment and valuation of ecosystem services. A review of this debate and the

92 creation of an adapted framework for operationalizing the ecosystem services concept for soil
93 threats and land management lay the foundation for the project.

94 Despite various research activities around the world over the last decades, integrating the
95 concept of ecosystem services into land management decision-making continues to pose
96 considerable challenges, and a coherent approach to assessing and valuing ecosystem services
97 is still lacking (de Groot et al., 2010). Many different frameworks have been developed to
98 operationalize the concept, but these are often specific to a certain issue (e.g. biodiversity,
99 water) or level (e.g. national) and each have their own definitions and understandings of
100 particular terms. The task of an ecosystem services framework is to aid the identification of
101 services, as well as their role, values, and trade-offs therein, in order to inform policy and land
102 management decisions. This article reviews existing frameworks and approaches and
103 proposes an adapted framework for soil-related ecosystem services that is suited for practical
104 application in the prevention and remediation of soil degradation across Europe. After briefly
105 introducing the emergence of the ecosystem services concept, we review and compare
106 existing ecosystem services frameworks and evaluate their concepts and terminologies
107 (Section 2). Section 3 focuses on soil aspects and on the contradictory use of soil functions
108 versus ecosystem services, while reviewing the current state of the art and identifying
109 knowledge gaps. We then evaluate existing approaches to monitor and value ecosystem
110 services (Sections 4 and 5, respectively). Furthermore, we examine how the ecosystem
111 services concept has been operationalized in research projects and land management in
112 Europe so far (Section 6). Based on our review, we develop a framework for considering soil
113 ecosystem services that is applicable to all soil threats and land management contexts
114 (Sections 7 and 8), and reflect on how to operationalize this framework for practical
115 application, particularly to support decision-making in preventing and remediating soil

116 degradation in Europe (Section 9). We conclude with an outlook on how the new framework
117 could support ongoing global efforts (Section 10).

118

119 **2. Comparing ecosystem services frameworks**

120 The ecosystem services concept is considered a useful tool to communicate and highlight the
121 dependence of human well-being on ecosystems. It has the potential to bridge the gaps
122 between ecological, economic, and social perspectives and enable sustainable resource
123 management (Braat and de Groot, 2012). Its most recent definition as proposed by Braat and
124 de Groot (2012, p. 5) states that ‘Ecosystem services are the direct and indirect (flux of)
125 contributions of ecosystems to human well-being.’ The term ‘ecosystem services’ was first
126 proposed in the early 1980s to increase public awareness about the negative consequences of
127 biodiversity loss on human well-being (Ehrlich and Ehrlich, 1981; Mooney and Ehrlich,
128 1997).

129 Since the 1990s, the number of scientific papers addressing ecosystem services has increased
130 exponentially (Vihervaara et al., 2010), with the focus expanding to include natural capital
131 beyond biodiversity (Fisher et al., 2009). Economists recognized that ecosystems’
132 contributions to human well-being were more wide-ranging than previously thought and thus
133 heavily undervalued in decision-making (Braat and de Groot, 2012).

134 The release of the Millennium Ecosystem Assessment (MEA) (2003, 2005) finally led to
135 broad recognition of the need to integrate ecosystem services in policy decision-making
136 (Gómez-Baggethun et al., 2010). The potential of an ecosystem for providing ecosystem
137 services depends on ecosystem functioning, which in turn depends on the ecosystem’s

138 biophysical structure (of which soils are a part) and on ecosystem processes (de Groot et al.,
139 2010). The MEA defines four types of ecosystem services as summarized below:

140 (1) *Provisioning services*: products obtained from ecosystems, including food, fibre, fuel,
141 land, water, medicinal, biochemical, genetic, and ornamental resources.

142 (2) *Regulating services*: benefits obtained from the regulation of ecosystem processes,
143 including carbon sequestration, erosion control, flood protection, pollination, water
144 purification, and waste management.

145 (3) *Cultural services*: non-material benefits that individuals obtain from ecosystems (through
146 use and non-use), including spiritual, religious, and cultural heritage, as well as
147 recreation, tourism, landscape, and amenity.

148 (4) *Supporting services*: services that are necessary for the production of all other ecosystem
149 services, such as soil formation and retention, cycling processes, and habitat provision.

150 The identification and assessment of processes driving the degradation of ecosystem services
151 directly (land use change, climate change, spread of exotic species, contamination, etc.) or
152 indirectly (demographic change, socio-economic change, etc.) were recommended as a basis
153 for decision-making (MEA, 2005).

154 Critics of the MEA's approach state that this classification mixes processes for achieving
155 services (means) and the services themselves (ends) in the same categories; for example,
156 water regulation is a process to achieve potable water (Wallace, 2007). To achieve practical
157 applicability, operationalization frameworks need to distinguish between intermediate
158 services (e.g. water regulation), final services (e.g. provision of clean water), and benefits
159 (e.g. drinking water) (Boyd and Banzhaf, 2007; Fisher et al., 2009). In response to these
160 criticisms, another large collaborative initiative, The Economics of Ecosystems and
161 Biodiversity (TEEB) (TEEB, 2010), developed a new cascading framework that distinguishes

162 between the biophysical structure, functions, services, benefits, and values (Figure 1). It was
163 supported by the United Nations Environment Programme (UNEP) and the European
164 Commission and many experts currently consider it the best available framework for
165 ecologically-based social and economic decision-making (Braat and de Groot, 2012).

166

167 [Figure 1 approximately here]

168

169 TEEB recommends three steps to analyse and structure ecosystem valuation: 1) Identify and
170 assess the full range of ecosystem services; 2) Estimate and demonstrate the value of
171 ecosystem services; 3) Inventory and manage the values of ecosystem services and seek
172 solutions to overcome their undervaluation. In a recent report about different approaches to
173 value ecosystem services in Europe, Brouwer et al. (2013) concluded that ‘one of the main
174 findings is that there does not exist one single, standard “TEEB” method or approach’ (p. 5).
175 To reach the target set by the EU 2020 Biodiversity Strategy of valuating ecosystem services
176 in Europe, the existing frameworks need to be further integrated and implemented (Brouwer
177 et al., 2013).

178 Further clarification of existing ecosystem services frameworks is offered by the Common
179 International Classification of Ecosystem Services (CICES) initiative, which developed from
180 work on environmental accounting undertaken by the European Environment Agency (EEA)
181 (Haines-Young and Potschin, 2013). The CICES views ecosystem services as arising from the
182 interaction of biotic and abiotic processes, and refers specifically to the ‘final’ outputs or
183 products from ecological systems – that is, the goods or services directly consumed or used by
184 people. Following TEEB, the CICES recognizes these outputs as provisioning, regulating, and
185 cultural services; it does not, however, cover the so-called ‘supporting services’ defined in the
186 MEA. Instead, these are treated as part of the ecosystem’s underlying structures and

187 processes. This change of perspective is particularly relevant to soils and soil processes, given
188 that the MEA classified them as supporting services. The latest version of the CICES (V4) has
189 a five-level hierarchical structure consisting of sections, divisions, groups, classes, and class
190 types. The highest level consists of the three familiar sections adopted from the MEA (see
191 CICES V4, www.cices.eu). The CICES has the disadvantage of being based mainly in the
192 natural sciences, leading to weak inclusion of social aspects, and it has become rather
193 complex, with extensive use of specialized terminology. Nonetheless, it has contributed
194 considerably to standardized naming of ecosystem services. The CICES also links up with
195 efforts to determine standards in environmental accounting and to integrate ecosystem
196 services into national accounting systems such as the System of Environmental-Economic
197 Accounting (SEEA) (Edens and Hein, 2013).

198 The MEA, TEEB, the CICES, and subsequent initiatives have tried to clarify the jumble of
199 terms in ecosystem services frameworks. Despite these efforts, a clear and generally accepted
200 framework and agreement on terms is still lacking. For example, what TEEB refers to as an
201 ecosystem's 'biophysical structure' is often called 'biophysical process' or 'biophysical
202 property' by other initiatives (Braat and de Groot, 2012; Maes et al., 2012; Müller and
203 Burkhard, 2012; and others). Together with the ecosystem functions it supports or provides,
204 this ecosystem side of the framework has also been named 'natural capital stocks' (Dominati
205 et al., 2010) or 'ecosystem potential' (Bastian et al., 2013; Haines-Young et al., 2012; Rutgers
206 et al., 2012). On the human well-being side of the framework, TEEB suggests distinguishing
207 between 'services', 'benefits' and (economic) 'value', while others refer to 'intermediate
208 services' and 'final services' (Crossman et al., 2013) and highlight the distinction of services
209 supply and demand. Some authors describe the 'services' in TEEB as 'provision', and
210 'benefits' as 'use/services', while 'value' is referred to as 'the importance or appreciation of a

211 service'. This lack of a consistent typology and terminology has led to numerous terms – such
212 as properties, processes, functions and services – being used interchangeably (Robinson et al.,
213 2013). Without clarifying these terms and concepts, we risk losing sight of the basic premise
214 of considering natural capital and processes separately from the services they support. One of
215 the results of this review is thus the development of a framework with clearly defined and
216 consistently used terms (see Section 7).

217 **3. Soil functions and ecosystem services**

218 Soil scientists have recently recognized the importance of the ecosystem services concept for
219 the prevention and mitigation of soil degradation (Bouma, 2014). A focus on soils requires
220 differentiating ecosystem services delivered specifically by soils from those that are provided
221 more generally by land (of which soil is a part). To date, soil ecosystem services have often
222 been valued only implicitly within those of land (Robinson et al., 2014). The ecosystem
223 services concept legitimates soil conservation practices by illustrating the broad value of
224 healthy soils, and it aids their evaluation regarding trade-offs. This insight has spurred efforts
225 to incorporate the ecosystem services concept in soil policymaking (Breure et al., 2012;
226 Robinson et al., 2012).

227 Within the soil science community, the ecosystem services framework is often used in
228 conjunction with the concept of soil functions. This latter concept emerged in Europe in the
229 early 1970s (Glenk et al., 2012) and was adopted to develop a proposal for the EU Soil
230 Framework Directive, highlighting seven key soil functions (European Commission, 2006):

- 231 • Biomass production, including in agriculture and forestry
- 232 • Storing, filtering, and transforming nutrients, substances, and water
- 233 • Biodiversity pool, such as habitats, species, and genes
- 234 • Physical and cultural environment for humans and human activities

- 235 • Source of raw materials
- 236 • Acting as carbon pool (store and sink)
- 237 • Archive of geological and archaeological heritage

238 However, the soil functions concept exists in many different forms. Blum (2005) categorized
239 soil functions into ‘ecological functions’ and ‘non-ecological functions’, with ecological
240 functions consisting of ‘biomass production’, ‘protection of humans and the environment’,
241 and ‘gene reservoir’. Non-ecological functions include ‘physical basis of human activities’,
242 ‘source of raw materials’ and ‘geogenic and cultural heritage’.

243

244 Soil functions are often used interchangeably with soil roles and soil ecosystem services,
245 leading to different combinations of categories across the various lists. According to Jax
246 (2005), the term ‘function’ is used in four main ways (see Glenk et al., 2012, p. 10):

- 247 • as a synonym for ‘process’;
- 248 • referring to the operation or function(ing) of a system;
- 249 • as a synonym for ‘role’; and
- 250 • as a synonym for ‘service’.

251

252 In order to avoid confusion with the well-understood term of soil processes, we suggest using
253 ‘soil function’ in the sense of ‘soil role’. The role or function of soils is to provide
254 (ecosystem) services. Soil processes, by contrast, support this provision of ecosystem services
255 and thus represent the capacity of an ecosystem to supply provisioning, regulating, and
256 cultural services.

257 Dominati et al. (2010) pointed out that the existing literature on ecosystem services tends to
258 focus exclusively on ecosystem services rather than holistically linking these services to the

259 natural capital base in which they originate. Although soils are major suppliers of critical
260 ecosystem services, soil-related ecosystem services are often not recognized, not well
261 understood, and thus not incorporated into the ecosystem services frameworks. As a result,
262 the link between soil natural capital and these ecosystem services is generally overlooked
263 (Breure et al., 2012). Haygarth and Ritz (2009) suggested combining ecosystem services with
264 soil functions that are relevant to soils and land use in the UK. They paired each of their
265 identified 18 services with a related soil function. Dominati et al. (2010, p. 1860) suggested
266 the following roles of soils in the provision of services:

- 267 • Fertility role
- 268 • Filter and reservoir role
- 269 • Structural role (i.e. physical support)
- 270 • Climate regulation role
- 271 • Biodiversity conservation role
- 272 • Resource role

273 These correspond roughly to the soil functions as presented by the European Commission
274 (2006) above, and, in our view, overlap with what is generally considered ecosystem services.
275 One aspect that might be added is the increasing awareness of cultural services.

276 Glenk et al. (2012) considered the following frameworks as the most comprehensive and as
277 the ones most consistently classifying and describing the linkages between soil and its
278 management and resulting impacts on ecosystem services: the ones proposed by Robinson
279 and Lebron (2010), Dominati et al. (2010) and Bennett et al. (2010). Glenk et al.'s key
280 message is that 'soil functions should be viewed as (bundles of) soil processes that are
281 providing input into the delivery of (valued) final ecosystem services' (p. 35). Robinson et al.
282 (2013) propose an earth system approach to provide more visibility to soils and other

283 compartments of the earth system in the supply chain for ecosystem services. Although this
284 approach includes many valuable considerations and a useful focus on soils, its stock–flow
285 model becomes rather complex for practical application.

286 **4. Measuring, Monitoring, and mapping ecosystem services**

287 Ecosystem services researchers have undertaken major efforts to quantify and measure
288 ecosystem services. Considerable effort has been put into identifying the relevant indicators
289 and ways of measuring them in order to map and quantify ecosystem services at different
290 spatial and temporal scales. Changes in ecosystem services need to be identified and
291 quantified as comprehensively as possible. The exclusion of some classes of services just
292 because they are difficult to quantify and measure must be avoided (Braat and de Groot,
293 2012). Quantifying bundles of ecosystem services and recognizing interrelations between
294 individual indicators within indicator sets, however, remains a major challenge when it comes
295 to monitoring ecosystem services flows.

296 Müller and Burkhard (2012) made various suggestions on how to raise indicator quality, such
297 as improving knowledge about relevant causal relations, recognizing interrelations between
298 indicators, improving the transparency of indicator derivation strategies, finding case-specific
299 optimal degrees of indicator aggregation, assessing indicator uncertainties, or estimating
300 normative loading in the indicator set. Specific indicators are needed for each component of
301 the ecosystem services framework. On the ecosystem side, property and function indicators –
302 also called state indicators – provide information about potential services of an ecosystem,
303 while on the human well-being side, performance indicators provide information about how
304 much of these potential services is actually provided and/or used (van Oudenhoven et al.,
305 2012).

306 A quantitative review of 153 regional ecosystem services case studies by Seppelt et al. (2011)
307 concluded by highlighting four aspects that would help to ensure the scientific quality and
308 holistic approach of further ecosystem services studies: (1) biophysical realism of ecosystem
309 data and models; (2) consideration of local trade-offs; (3) recognition of off-site effects (i.e.
310 ecosystem services provision at different scales); and (4) comprehensive but critical
311 involvement of stakeholders in assessment studies. The holistic involvement of a variety of
312 stakeholders makes it possible to assess who has what ability to benefit from services. This is
313 important because trade-offs occur not only between services (Viglizzo et al., 2012) but also
314 between beneficiaries (Horcea-Milcu et al., 2016).

315 A huge amount of research has focused on mapping ecosystem services, and the variety of
316 approaches has triggered several review papers on the methodologies used (e.g. Burkhard et
317 al., 2009; Eigenbrod et al., 2010; Maes et al., 2012; Crossman et al., 2013). Maes et al. (2012)
318 found that provisioning ecosystem services can be mapped and quantified easily and directly,
319 whereas most regulating, supporting, and cultural services are more difficult to locate and
320 require proxies for their quantification. Additionally, these authors point out that the
321 connection between the status of an ecosystem and the services it delivers is still poorly
322 explored. This is particularly critical with regard to soil-related services, as soil status can be
323 masked for a certain time (e.g. using fertilizer).

324 Most mapping approaches are applied at national or even continental scales, and they are
325 mainly used to support decision-making on changes in land use rather than land management.
326 However, adapting land management is often more feasible and hence more effective in
327 mitigating soil threats than completely changing the land use.

328 Only few studies have quantified and measured ecosystem services specifically related to soil;
329 among them are studies by Rutgers et al. (2012), Schulte et al. (2014), and Dominati et al.

330 (2014). A preliminary method for the quantification of soil quality indicators on arable farms
331 was developed by Rutgers et al. (2012). These researchers had land users and experts score
332 various ecosystem service indicators for their importance and informative value and then
333 calculated a final indicative score for each indicator. This process should not be confused with
334 ecosystem services valuation (see Section 5), as it represents a preliminary step before
335 assessing actual service provision (which in turn might be compared to a maximum ecological
336 potential, resulting in a performance index, as done by Rutgers et al., 2012). Another effort to
337 develop a method for the quantification of soil-related ecosystem services was undertaken by
338 Dominati et al. (2014), who worked with a comprehensive list of proxies for each service and
339 units for measuring them. This study omitted cultural services due to their non-biophysical
340 nature and the related challenges of quantifying them. The use of proxies is often inevitable
341 due to the complexity and number of ecosystem services, but it requires careful consideration.
342 Eigenbrod et al. (2010) compared primary data for biodiversity, recreation, and carbon storage
343 in the UK with land-cover-based proxies and found a poor data fit and potentially large errors
344 associated with proxy data. They recommend investing in survey efforts rather than using
345 poor-quality proxy data, and conclude that surveys can be more cost-effective in the end.

346 Agriculture and land management can have a direct influence on ecosystem properties,
347 functions, and services. Van Oudenhoven et al. (2012) applied the stepwise cascade model
348 proposed by Haines-Young and Potschin (2010) to a multifunctional rural landscape in the
349 Netherlands, assessing land management effects without confusing ecosystem properties,
350 functions, and services, and thus avoiding double-counting. They confirmed that function
351 indicators are a ‘subset or combination of ecosystem property indicators, as was earlier
352 suggested by Kienast et al. (2009)’ (van Oudenhoven et al., 2012, p. 118). Differences in
353 ecosystem services between land management systems offer potential for mitigating trade-

354 offs by combining contrasting services in strategically designed landscape mosaics (Lavelle et
355 al., 2014).

356 Due to methodological challenges, cultural ecosystem services are generally only roughly
357 included in ecosystem services assessments. At the same time, many authors clearly underline
358 the importance of these immaterial benefits, especially those of cultural landscapes
359 (Plieninger et al., 2013; Chan et al., 2012; Paracchini et al., 2014). Plieninger et al. (2013)
360 stressed that spatially explicit information on cultural ecosystem services – as perceived by
361 the local population – provides the basis for developing sustainable land management
362 strategies, including biodiversity conservation and cultural heritage preservation. Work done
363 in the UK by Kenter et al. (2014) suggests that analysis of cultural ecosystem services can be
364 developed using quantitative indicators and drawing on publicly available datasets, such as
365 surveys of recreation usage. However, they also emphasize the importance of participatory
366 and interpretative research techniques developed in the social sciences to assess and
367 understand cultural ecosystem services in location- and community-based contexts.

368

369 **5. Valuing ecosystem services**

370 The ecosystem services concept is intrinsically connected to values. It aims to provide a link
371 between the supply of nature's goods and services and how they are valued by society.
372 Indeed, much emphasis has been placed on valuing ecosystem services, with the aim of
373 demonstrating that markets fail to adequately reflect the full value society gives to ecosystem
374 services and hence often co-drive the degradation of ecosystems. The large body of literature
375 on ecosystem services valuation has consistently shown that non-market values nearly always
376 outweigh market values (e.g. Ananda and Herath, 2003; Shiferaw and Holden, 1999),

377 although ways in which the latter are derived are often contested. Four research traditions
378 have investigated the valuation of ecosystem services to support better informed decisions:

- 379 1. One school stresses the need to convert all values in monetary figures. Although its
380 proponents are mindful of various shortcomings, their rationale is that decision- and
381 policymakers are more likely to appreciate the full value of nature if they are
382 confronted with a single figure indicating the total economic value of all services of an
383 ecosystem. Because such a figure is more difficult to provide for soils than for other
384 ecosystem components, the significance of soils is underplayed. Prominent examples
385 include Costanza et al.'s (1997, 2014) value of the earth's natural capital, as well as
386 TEEB's Ecosystem Service Valuation Database (de Groot et al., 2012; van der Ploeg
387 and de Groot, 2010).
- 388 2. A second school regards markets as inherently unsuitable for valuing nature, and
389 objects to expressing the value of ecosystems in monetary terms (e.g. Sagoff, 2008).
390 Proponents of this tradition hold that decisions must take account of different value
391 systems and multiple criteria for assessing value. Any attempt to express value in
392 monetary terms would reduce the dimensions considered, weakening the potential to
393 achieve sustainability (also referred to as 'weak sustainability', see e.g. Ayres et al.,
394 2001).
- 395 3. A third school focuses more on operational difficulties to maximize the value of
396 ecosystem services. Managing land to maximize one (bundle of) ecosystem services
397 often requires sacrificing value derived from other ecosystem services. The ecosystem
398 services concept is well-suited to studying such trade-offs between different
399 ecosystem services. An important initiative based on this paradigm is the Natural
400 Capital project with its InVEST methodology (Kareiva et al., 2011).

401 4. A fourth, emerging school has an even stronger focus on values rather than valuation,
402 and in this sense constitutes an extension of schools 2 and 3 above. In this school,
403 ecosystem services are seen as part of the social-ecological system (Folke, 2006;
404 Olsson et al., 2004). Values associated with ecological knowledge and understanding
405 play an important role in the provision of bundles of ecosystem services, as do the
406 social networks associated with them. They are considered important for developing
407 resilience within social-ecological systems and ecosystem services (CGIAR Research
408 Program on Water, Land and Ecosystems, 2014).

409 The valuation of ecosystem services is examined by a large body of ecological economics
410 literature. Economic valuation is based on an anthropocentric approach and defines value
411 based on individual preferences. This approach is typically taken by the first school described
412 above. The Total Economic Value (TEV) framework captures the benefits derived from
413 ecosystem services. The total economic value of any resource is the sum of use and non-use
414 values (Figure 2).

415 [Figure 2 approximately here]

416 ‘Use value’ involves interaction with the resource and is subdivided into ‘direct use value’
417 and ‘indirect use value’. Direct use value relates to the use of natural resources in a
418 consumptive (e.g. industrial water abstraction) or in a non-consumptive manner (e.g. tourism).
419 From an ecosystem services perspective, direct use value is often associated with provisioning
420 (e.g. agriculture) and cultural ecosystem services (e.g. recreation activity). Indirect use value
421 relates to the role of natural resources in providing or supporting key ecosystem services (e.g.
422 nutrient cycling, climate regulation, habitat provision). In ecosystem services terminology,
423 indirect use value is frequently attached to regulating ecosystem services.

424 'Non-use value' is associated with benefits derived from the knowledge that natural resources
425 and aspects of the natural environment are being maintained. Non-use value can be split into
426 two parts: (1) bequest value (associated with the knowledge that the area as a resource will be
427 passed on to future generations), and (2) existence value (derived from the satisfaction of
428 knowing that a resource continues to exist, regardless of use made of it now or in the future)
429 (Figure 2). Some authors have distinguished a third type of non-use value: (3) altruistic value
430 (derived from the knowledge that contemporaries can enjoy the goods and services related to
431 an area) (Hein, 2010; Kolstad, 2000). Option value can be both use or non-use value, and it is
432 not associated with current use of a resource but with the benefit of keeping open the option to
433 make use of it in the future. Within overall valuation of nature, the question of valid
434 components and methodologies for assessing non-use values has been particularly hotly
435 debated.

436 The available approaches and methods for ecosystem services valuation can be categorized as
437 follows: (1) direct market valuation approaches (e.g. approaches based on market price, costs,
438 or production function); (2) revealed-preference approaches (e.g. travel cost method, hedonic
439 pricing approach) and (3) stated-preference approaches (e.g. contingent valuation method,
440 choice experiment model, group valuation) (Chee, 2004; Pascual et al., 2010). Encompassing
441 the monetary values of ecosystem services provisioning in integrated economic tools such as
442 cost-benefit analysis and cost-effectiveness analysis can be very useful in evaluating policy
443 options (e.g. land management measures for prevention and restoration). However, the
444 methods outlined above have been criticized for being too hypothetical in complex situations
445 (Getzner et al., 2005). Efforts are now being made to develop more deliberative valuation
446 techniques that enable more open and potentially more grounded outputs in complex
447 situations by combining stated-preference approaches with increased deliberation between

448 experts and/or users. These techniques' outputs are more culturally constructed and richer
449 from a contextual point of view and potentially consider a wider range of ecosystem services
450 within any given valuation (Kenter et al, 2014).

451 **6. How have European research projects operationalized the soil ecosystem services** 452 **concept?**

453 A previous systematic review by Vihervaara et al. (2010) showed that in publications up to
454 2008, the ecosystem services concept had been underexplored in relation to soil quality and
455 regulation compared with biodiversity, and in agricultural systems compared with watersheds
456 and forestry. This can be explained by the concept's history (see Section 2). To assess more
457 recent developments and understand how the ecosystem services concept is being developed
458 in relation to soils, we did a rapid systematic review of current and recent (mainly post-2008)
459 soil research projects. To this end, we searched Scopus on 22 April 2014 for papers
460 containing the keywords 'ecosystem services' and 'soils'. The results were then narrowed
461 down to 1,137 publications that also contained the keyword 'Europe'. Using titles and
462 abstracts, the list was further narrowed down by excluding those that did not match the
463 combination of all three search criteria. The text and acknowledgments of the remaining 200
464 papers were then scanned for mention of the projects that supported or funded the research.
465 This resulted in a list of 50 projects. Exploring information available on the Internet, we
466 identified a number of project characteristics that could be used to categorize and compare the
467 projects; at the same time, we excluded a number of projects that did not meet the criteria or
468 for which no information was available. This resulted in a total of 39 projects being
469 categorized and compared (see Appendix A, Table A).

470 First, we categorized the projects according to how explicitly they addressed soil ecosystem
471 services. Only eight projects focused specifically on soil ecosystem services. Examples

472 include the SOIL SERVICE project that explicitly focuses on soil biodiversity, or SoilTrEc,
473 which focuses on soil processes in river catchments. The SmartSOIL project explicitly
474 examined soil ecosystem services driven by soil organic carbon (i.e. food production and
475 climate regulation). The project informed farmers, advisers, and policymakers about benefits,
476 drawbacks, and costs of land management practices that increase or sustain soil carbon.
477 Another 18 projects included soil ecosystem services more implicitly in their research,
478 considering them as intermediary services contributing to the ecosystem services on which the
479 projects mainly focused. Many of these projects (e.g. RUBICODE, MULTAGRI,
480 LIBERATION) focused on biodiversity and included soil in terms of its potential impact on
481 biodiversity and ecosystem services. 13 projects were categorized as hybrids somewhere in
482 between the above two categories. We found that projects focusing specifically on soils are
483 usually run by large consortia and funded by the European Commission or similar
484 international funding agencies. There were also a number of projects funded by national
485 agencies in an effort to establish research with a national focus (e.g. MOUNTLAND) or small
486 research centres (e.g. FuturES). These tended to have quite a broad ecosystem services focus
487 and were therefore attributed to the hybrid category.

488 Next, we categorized projects based on whether they focused more on baseline knowledge or
489 more on management impacts. Of the 39 projects, 34 were found to be 'baseline' projects that
490 seek to characterize ecosystem services and understand their relationships. They monitor
491 ecosystem services, observing changes or impacts of changes on benefits or on other
492 ecosystem services. Their aim is to build an understanding of which services exist, how they
493 are linked or bundled through benefits, and what trade-offs and gains result from the
494 prioritization of certain services. Much of the soil-focused research (including the work done
495 by the SOIL SERVICE project) falls into this category. Similarly, 30 out of the 39 projects

496 were categorized as ‘management’ projects that build on this baseline knowledge by studying
497 how management interventions impact on ecosystem services. Management interventions
498 usually involve physical changes, such the planting of trees to reduce erosion. ‘Management’
499 projects often contribute to ‘baseline’ projects by monitoring the ecosystem services affected
500 by the intervention being assessed. Most projects in this category focus on biodiversity (e.g.
501 MULTAGRI, AGFORWARD). They also predominantly focus on agricultural land and
502 hence implicitly include soil ecosystem services, although these are rarely specifically
503 examined.

504 Finally, we examined how closely projects were related to decision-making and
505 policymaking. We found that 23 projects can be characterized as decision-making or policy
506 research that seeks to aid the promotion of ‘successful’ ecosystem services management.
507 Many of these projects designed tools to support land use decision-making (e.g.
508 LandSFACTS); others proposed policy responses to promote the uptake of ecosystem services
509 management initiatives or to prevent damage to ecosystem services. A third subset in this
510 category consists of projects that explicitly seek to support payments for ecosystem services
511 by valuing these ecosystem services. Most projects in this subset do not have soil ecosystem
512 services as an explicit focus.

513 Regardless of whether projects focused on baseline or on management knowledge, or how
514 closely they were related to decision-making, the majority of projects focused on individual
515 ecosystem services or bundles of ecosystem services (e.g. those related to biodiversity). This
516 means that they zoomed in on components of the soil system. As a result, they were unable to
517 assess how the studied ecosystem services interacted with others in the context of a soil threat,
518 or to consider trade-offs between bundles of ecosystem services. A notable exception is the
519 SoilTrEC project, which takes a holistic approach to understanding soil processes in river

520 catchments. The project notes the need for ‘a clear operational framework to convey soils
521 research within the ecosystem services approach’ (Robinson et al., 2013 p. 1032).

522 The baseline knowledge which is being generated by current projects provides empirical data
523 on individual, or groups of, ecosystem services. It thus provides a useful basis for the
524 subsequent development of management and policy approaches. Moreover, this baseline
525 knowledge is supplemented by research that implicitly focuses on soil ecosystem services as
526 intermediary services contributing to end services such as water regulation. However, there
527 remains a research and conceptual gap in relation to fully operationalizing ecosystem services
528 for the mitigation of soil threats. Aiming to fill this gap within the RECARE project, we have
529 developed an adapted ecosystem services framework, which is outlined in the next section.

530 **7 Requirements of an adapted framework to operationalize ecosystem services for the** 531 **mitigation of soil threats**

532 Although many ecosystem services frameworks have been developed over time, choosing one
533 that is appropriate to operationalize ecosystem services for the mitigation of soil threats
534 remains challenging. RECARE aims to assess, at various spatial scales, how soil processes
535 and ecosystem services are affected by soil threats and by prevention and remediation
536 measures. We plan to use the ecosystem services concept for communication with local
537 stakeholders to identify the most beneficial land management measures, and with national and
538 European policymakers to identify trade-offs and win–win situations resulting from, and/or
539 impacted by, European policies. The chosen framework must therefore reflect and
540 acknowledge the specific contributions of soils to ecosystem services, and it must be capable
541 of distinguishing changes in ecosystem services due to soil management and policies
542 impacting on soil. At the same time, it must be simple and robust enough for practical
543 application with stakeholders at various levels. Our literature review and feedback from

544 scientists and policymakers at various conferences clearly showed that there is a need for (1) a
545 framework that focuses specifically on soil ecosystem services, (2) clarification of the terms
546 used therein, and (3) practical applicability of this framework.

547 Our review of ecosystem services frameworks revealed that none of the existing frameworks
548 fully suits these requirements. We identified three major challenges that need to be addressed
549 when working with, and thus adapting, an ecosystem services framework within the RECARE
550 project (as well as beyond):

- 551 • Linking ecosystem services to soils as well as to land management
- 552 • Ensuring that the framework can be used with stakeholders at various scales to assess
553 and value services provided by soils and affected by land management (to mitigate soil
554 threats)
- 555 • Ensuring that the framework is both scientifically robust and simple

556 These challenges outline the research gap which this paper aims to close by adapting existing
557 ecosystem services frameworks. We started from the framework proposed by Braat and de
558 Groot (2012), which we sought to complement with elements from more soil-specific recent
559 suggestions, for example by Dominati et al. (2014) while attempting to introduce a consistent
560 terminology that is understandable to a variety of stakeholders. This is in line with
561 suggestions by authors such as Bouma, who stated that achievement of the UN Sustainable
562 Development Goals will require more effective use of transdisciplinary approaches by soil
563 scientists (Bouma, 2014). The adapted ecosystem services framework, presented in Figure 3,
564 uses the following elements from existing frameworks:

- 565 • MEA (2005): major categories of ecosystem services
- 566 • TEEB (2010): subcategories of ecosystem services, but adapted and simplified

- 567 • Haines-Young and Potschin (2010): cascade model
- 568 • Braat and de Groot (2012): main model structure and feedback loops in TEEB model
- 569 • SmartSOIL (Glenk et al., 2012): soil processes, benefits
- 570 • Van Oudenhoven et al. (2012): land management, driving forces, societal response
- 571 • Dominati et al. (2014): natural capital, with inherent and manageable properties of
- 572 soil; external drivers as ‘other driving forces’, degradation processes as ‘soil threats’
- 573 • CICES (2013) and Mapping and Assessment of Ecosystems and their Services
- 574 (MAES) (Maes et al., 2013) were considered, but without taking elements.

575 [Figure 3 approximately here]

576

577 **8 The RECARE ecosystem services framework**

578 Like many other ecosystem services frameworks, the RECARE framework distinguishes
579 between an ecosystem side and a human well-being side. Given that the RECARE project
580 focuses on soil threats, soil threats are the starting point on the ecosystem side of the
581 framework. Soil threats affect natural capital such as soil, water, vegetation, air, and animals,
582 and are in turn influenced by these. Within the natural capital, the RECARE framework
583 focuses in particular on soil and its properties, which it classifies into ‘inherent’ and
584 ‘manageable’ properties. According to Dominati et al. (2014), inherent properties include
585 slope, orientation, depth, clay types, texture, size of aggregates (subsoil), stoniness, strength
586 (subsoil), subsoil pans, and subsoil wetness class; manageable properties include soluble
587 phosphate, mineral nitrogen, soil organic matter, carbon content, temperature, pH, land cover,
588 macroporosity, bulk density, strength (topsoil), and size of aggregates (topsoil). However, this
589 distinction between inherent and manageable soil properties is arguable: for example,
590 stoniness and wetness class are simultaneously inherent and manageable, as stones can be

591 removed and wetness influenced; whereas some of the subsoil properties may only change
592 after decades of management and are thus considered to be more clearly inherent. Similarly,
593 temperature, bulk density, strength, and size of aggregates can theoretically be influenced by
594 man, but are in practice difficult to manage. A number of these properties could thus be
595 exchanged between the two lists presented in Table 1. This also depends on the type of soil
596 being assessed and on its vertical structure, so a valid distinction might only be possible
597 within a local context.

598 [Table 1 approximately here]

599 Water, vegetation, and animal properties, in particular, are mostly manageable and have a
600 considerable influence on soil processes and ecosystem services. Air influences soil processes
601 through the exchange of gases and fine particles and is linked to soil threats through airborne
602 pollutants and the direct emission from and/or capturing of greenhouses gases in soils. Air can
603 be managed by adapting the land cover, land use, and land management. Some of these non-
604 soil properties are also listed in Table 1, but the list is certainly not yet exhaustive.
605 Application of the framework within RECARE will provide an opportunity for completing
606 and refining the property lists.

607 The natural capital's properties enable or influence soil processes, while at the same time
608 being affected by them. Soil processes represent the ecosystem's capacity to provide services;
609 that is, they support the provision of ecosystem services. Because we consider soil functions
610 to be synonymous with ecosystem services, we decided to omit the former term from our
611 framework. This will help to avoid confusion among readers associating the term with a
612 different meaning (see Section 3).

613 'Provisioning services' include biomass production, water production, the supply of raw
614 materials, and the physical base; 'regulating and maintenance services' include air quality
615 regulation, waste treatment, water regulation and retention, climate regulation, maintenance of
616 soil fertility, erosion control, pollination, biological control, lifecycle maintenance, habitat,
617 and gene pool protection; and 'cultural services' include the enabling of spiritual and aesthetic
618 experiences, the provision of inspiration, and the representation of cultural heritage.

619 Ecosystem services may be utilized to produce benefits for individuals and the human society,
620 such as food, drinking water, or hazard regulation. These benefits are explicitly or implicitly
621 valued by individuals and society. The monetary and intrinsic values attached to these
622 benefits can influence decision- and policymaking at different scales, potentially leading to a
623 societal response. A deliberative process of negotiating different policy priorities within a
624 multi-stakeholder forum makes it possible to achieve optimal societal value and sustainability.
625 Individual (e.g. farmers') and societal decision- and policymaking strongly determine land
626 management, which again affects soil threats and natural capital. Land management includes
627 physical practices in the field (i.e. technologies), but also the ways and means (e.g. financial,
628 material, legislative, educational) to implement these (i.e. approaches) (Liniger and Critchley,
629 2007; Schwilch et al., 2011). Technologies entail agronomic (e.g. no-till, intercropping),
630 vegetative (e.g. tree planting, grass strips), structural (e.g. terraces, dams) or management
631 measures (e.g. land use change, area closure, rotational grazing) that control soil and land
632 degradation and enhance productivity. These measures are often combined to reinforce each
633 other.

634 Red arrows in Figure 3 represent the key links relevant to soil threats and soil management
635 decision-making. These links are the main focus of RECARE, the aim being to operationalize

636 the ecosystem services concept for practical application in preventing and remediating
637 degradation of soils in Europe through land care.

638 The RECARE framework can be illustrated by the following example, which will help
639 readers understand the ideas behind the boxes and arrows in Figure 3: A land user's intensive
640 ploughing (*land management*) of sloping land under conditions of increasingly erratic rainfall
641 due to climate change, market pressure to produce more and at a predefined time, and the
642 tradition of preparing a fine seedbed (*other natural and human driving forces*) causes soil
643 erosion (*soil threat*). Among other things, this leads to reduced soil organic matter content in
644 the topsoil, changed topsoil aggregates, and reduced soil cover (*properties of the natural*
645 *capital*), which affects soil organic matter cycling, soil structure maintenance, and water
646 cycling (*soil processes*). This may result in reduced production of biomass and reduced off-
647 site water regulation (*ecosystem services*), causing a decline in yield and downstream flooding
648 (*benefits*). The loss in crop production and the downstream damage are given a negative value
649 by society, producers, and policymakers (*value*). This could be discussed in a multi-
650 stakeholder deliberation process and result in incentives for good agricultural practice
651 provided to land users by large agri-food corporates and/or the adjustment, improvement, or
652 more effective implementation of policies to protect soil against erosion and maintain key
653 ecosystem services (*decision- and policymaking*). This leads the land user to implement a no-
654 till practice (*land management*), which enhances soil organic matter, improves soil structure
655 and cover, and thus successfully combats soil erosion (*soil threat*). From here we can go
656 through the same parts of the framework again, which are now influenced in a positive way.
657 However, it is important to take into account trade-offs. In this example, the implemented no-
658 till practice might increase soil pollution owing to the application of herbicide, leading to a
659 trade-off between soil threats. Ideally, sustainable land management should simultaneously be

660 the starting point in the framework and the main aim of its application. Ultimately, the aim of
661 sustainable land management could imply taking precautionary measures to prevent soil
662 threats from even emerging.

663 The RECARE framework also relates to the DPSIR framework (Smeets and Weterings, 1999)
664 by viewing the *driving forces* ('driver'), including *land management*, as exerting 'pressure' on
665 soil resources, manifested through *soil threats*. These change the *properties of the natural*
666 *capital* ('status') and affect *ecosystem services* ('impact 1') and *human well-being* ('impact
667 2'). In response to both of these, society either changes its *decision- and policymaking*, or
668 land users directly adapt their *land management* ('response'), depending on their willingness
669 and ability. See also the article by Müller and Burkhard (2012), who suggest a similar link
670 between the ecosystem services and DPSIR frameworks from an indicator-based perspective.

671 Stakeholders can only improve ecosystem services through land management if these services
672 are 'manageable' for them. A small study in Australia assessed farmers' perceived ability to
673 manage ecosystem services (Smith and Sullivan, 2014). Only soil health and shade/shelter
674 were indicated as being highly manageable, with a high convergence in views. While
675 shade/shelter was a specific issue of the area, soil health was the only ecosystem service for
676 which farmers indicated being both highly vulnerable to its loss and able to influence it
677 themselves.

678 Measuring desired and achieved improvements in ecosystem services and in their underlying
679 soil processes requires the definition of indicators. A thorough review undertaken for the
680 RECARE project (Stolte et al., 2016) presents indicators for each soil threat. These enable
681 measuring the effects of soil threats and remediation measures based on key soil properties as
682 well as biophysical (e.g. reduced soil loss) and socio-economic (e.g. reduced workload)
683 impact indicators. In order for these indicators to be of use in operationalizing the ecosystem

684 services framework, it has to be possible to associate changes in their values (i.e. in soil
685 properties and processes) to impacts of prevention and remediation measures. This requires
686 the indicators to be sensitive to small changes, but still sufficiently robust to prove changes
687 and enable their association to land management.

688

689 **9. Operationalizing the RECARE ecosystem services framework**

690 The proposed new framework's output and the ways in which it can be put to use for
691 decision-making at various spatial scales will be further developed during the next years of
692 the RECARE project. The 17 RECARE case study sites across Europe with their diversity of
693 soil threats and land use systems will serve as a laboratory for operationalizing the
694 framework. Prevention and remediation measures were selected and are now being trialled in
695 all case study sites, and the changes in manageable soil and other natural capital properties are
696 being measured and quantified. An assessment of changes in soil processes and ecosystem
697 services based on meaningful aggregation and inclusion of proxy indicators will provide a
698 comprehensive appraisal of each measure's impact. This will include measurement of cultural
699 ecosystem services, which have largely been under-represented in ecosystem services
700 assessments so far. In order to guarantee practical applicability in decision-making, data
701 collection will be limited to the information needed to assess the measures' impacts. Evidence
702 from these impact assessments will then feed into stakeholder assessments. Stakeholders will
703 value the interventions' impacts on ecosystem services and then discuss and reflect on the
704 methods and policy recommendations.

705 So far, researchers from all study sites have drafted examples of potential outcomes for their
706 respective site. These include preliminary lists of expected changes in soil properties, affected
707 soil processes, and their assumed impacts on ecosystem services for the different soil threats

708 and prevention and remediation measures. Some consideration was also given to how the
709 framework can be embedded into existing and new governance structures. Two examples are
710 included here to illustrate the framework's operationalization: In the case of soil erosion as a
711 result of degradation and abandonment of agricultural terraces in Cyprus, an interdisciplinary
712 group of experts found that measures such as terrace rehabilitation, crop diversification,
713 afforestation, and improved design and management of unpaved roads could affect a variety
714 of ecosystem services. These services include water availability and quality (for households
715 and irrigation), erosion regulation, flood prevention, hazard regulation, soil formation, cultural
716 heritage, and recreation and tourism. The impacts arising from the selected land management
717 options, together with the perceived importance of each service, form the basis for
718 stakeholders' upcoming valuation of the relevant services and will lead to the evaluation of
719 land management practices and the formulation of policy advice. At another site, in the
720 Netherlands, dairy farmers created a foundation to finance and exchange knowledge on crop
721 and soil management practices that maintain or increase soil organic matter. They found that
722 undersowing of grass in maize fields resulted in improved root biomass and soil water holding
723 capacity.

724 The ecosystem services provided and influenced by prevention and remediation measures are
725 valued differently by different stakeholders. For this reason, RECARE aims to develop a
726 methodology that enables stakeholders at the local and (sub-)national levels to determine and
727 negotiate values in a deliberative process that is suitable for being embedded in local
728 governance structures. Based on our review, we envisage using stated preference methods –
729 namely, contingent valuation – to elicit stakeholders' willingness to pay for the specified
730 environmental changes, along with direct market valuation approaches. Cost-benefit analysis
731 will be applied to assess whether a prevention measure is likely to be adopted and to inform
732 policymaking. Other methods may be added following further assessment of existing

733 valuation tools (for monetary and non-monetary valuation) and their suitability for adaptation
734 to soil threat mitigation.

735 The main aim is to create a practical basis for decision support in soil management, which can
736 be used by local stakeholders, such as land users, river catchment groups, advisory services,
737 or companies, to select optimally suited soil management measures, and by local, regional,
738 national, and supranational planners and private-sector actors to shape investments, public–
739 private agreements, legislation, regulation policies, and subsidy schemes. The framework will
740 also be used as a basis to develop an integrated model for assessing the impact of different
741 planning and policy options on ecosystem services under various external conditions at
742 different scales. To ensure scalability, ecosystem service assessments will be scaled up from
743 the local to the regional, national, and supranational (European) levels using integrated
744 assessment modelling approaches (van Delden et al., 2011, 2010) that enable cost-
745 effectiveness and cost–benefit analyses of land management measures, approaches, and
746 policies (Fleskens et al., 2014).

747

748 **10. Conclusions**

749 The need for a soil-focused ecosystem services framework has been confirmed by the newly
750 revised World Soil Charter (FAO, 2015), whose Principle #10 states: ‘Soil degradation
751 inherently reduces or eliminates soil functions and their ability to support ecosystem services
752 essential for human well-being. Minimizing or eliminating significant soil degradation is
753 essential to maintain the services provided by all soils and is substantially more cost-effective
754 than rehabilitating soils after degradation has occurred.’ The UN Food and Agriculture
755 Organization’s (FAO’s) new definition of sustainable soil management will also incorporate
756 the concept of ecosystem services. Moreover, the UN Sustainable Development Goals (SDGs)

757 lists, as Sustainable Development Goal #15, to ‘protect, restore and promote sustainable use
758 of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and
759 reverse land degradation and halt biodiversity loss’ (United Nations, 2015). Given this
760 widespread recognition that soils play a key role in terrestrial ecosystems, the development of
761 appropriate tools to promote sustainable soil management is more than timely. With the soil-
762 focused ecosystem services framework proposed in this paper we intend to make a practical
763 contribution.

764

765 **Acknowledgements**

766 The research leading to these results has received funding from the European Union Seventh
767 Framework Programme (FP7/2007-2013) under grant agreement No. 603498 (RE CARE
768 project). The authors wish to thank all partners of RE CARE for their useful feedback on the
769 suggested ecosystem services framework during and after the plenary meeting in Padova in
770 March 2015. We are also grateful for the inputs and feedback received during the RE CARE
771 ‘Soil Threats and Ecosystem Services’ workshop in Wageningen in May 2014 and the Global
772 Soil Week dialogue session in Berlin in April 2015. Finally, we thank Marlène Thibault of
773 CDE for editing this article and the two anonymous reviewers for their helpful comments.

774

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995

Properties of the natural capital		
Soil	Inherent	Slope Orientation Depth Clay types Texture Temperature Size of aggregates (subsoil) Strength (subsoil) Subsoil pans
	Manageable	Soluble phosphate Mineral nitrogen Soil organic matter Carbon content Soil moisture (topsoil) Subsoil wetness class pH Chemical quality Stoniness Cover (stones, litter, vegetation, etc.) Macroporosity Bulk density Strength (topsoil) Size of aggregates (topsoil)
Water	Manageable	Irrigation Drainage Groundwater depth Surface water/runoff Chemical quality
Vegetation	Manageable	Cover Vertical structure (e.g. multi-story) Horizontal structure (e.g. patchiness, strips) Species composition Soil flora
Animals	Manageable	Amount (grazing pressure) Type composition Soil fauna and microorganisms
Air	Inherent	Temperature Humidity
	Manageable	Chemical quality

997 *Table 1: Properties of the natural capital (in relation to soil management). This list is not*
 998 *exhaustive. Inherent and manageable soil properties adapted from Dominati et al. (2014).*

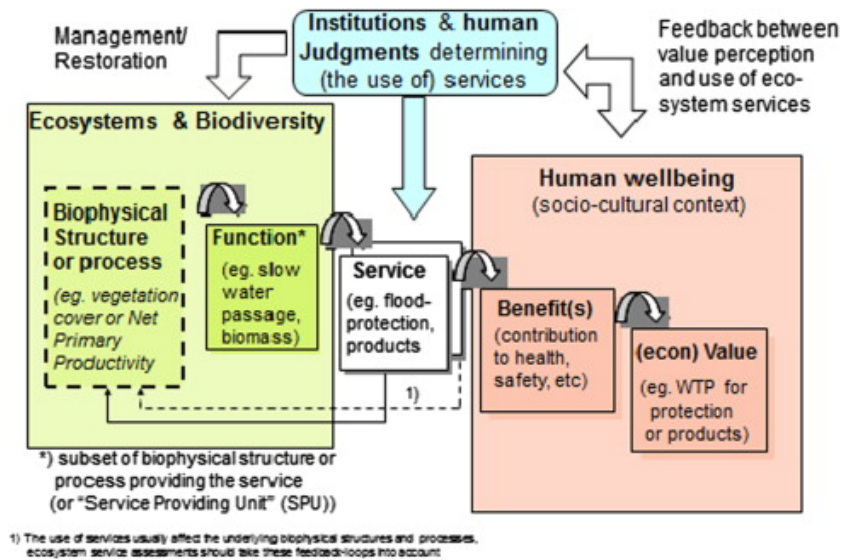
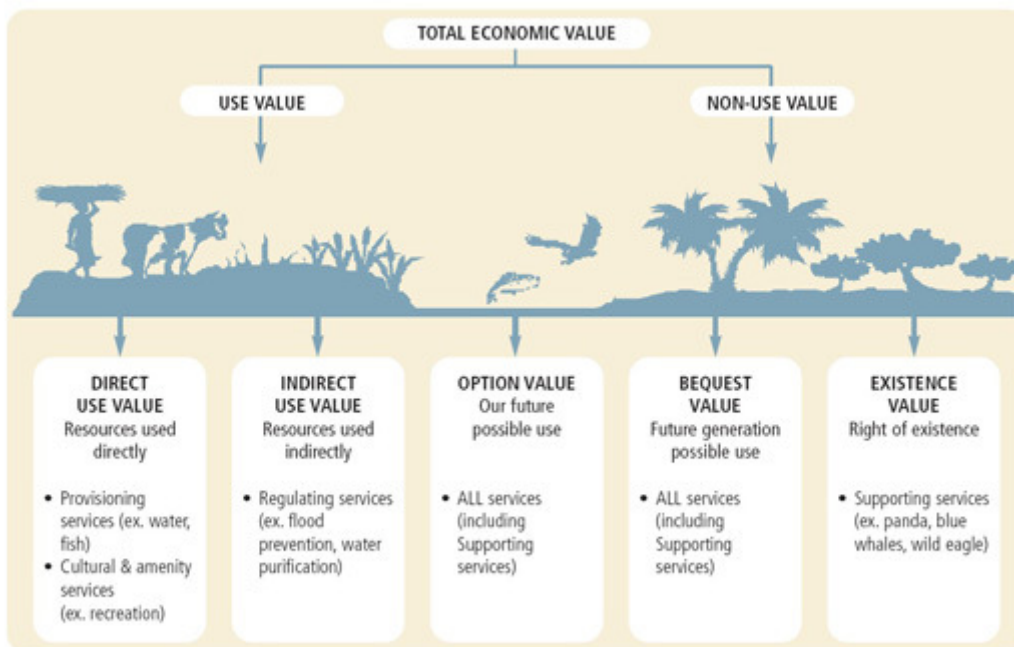


Figure 1: Overview of the framework developed by The Economics of Ecosystems and Biodiversity (TEEB). Designed for the purpose of economic valuation, this framework focuses mainly on economic values, without considering other value systems. Source: Braat and de Groot (2012), adapted from Haines-Young and Potschin (2009).

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Figure 2. Overview of the Total Economic Value (TEV) of ecosystems (Smith et al., 2006).

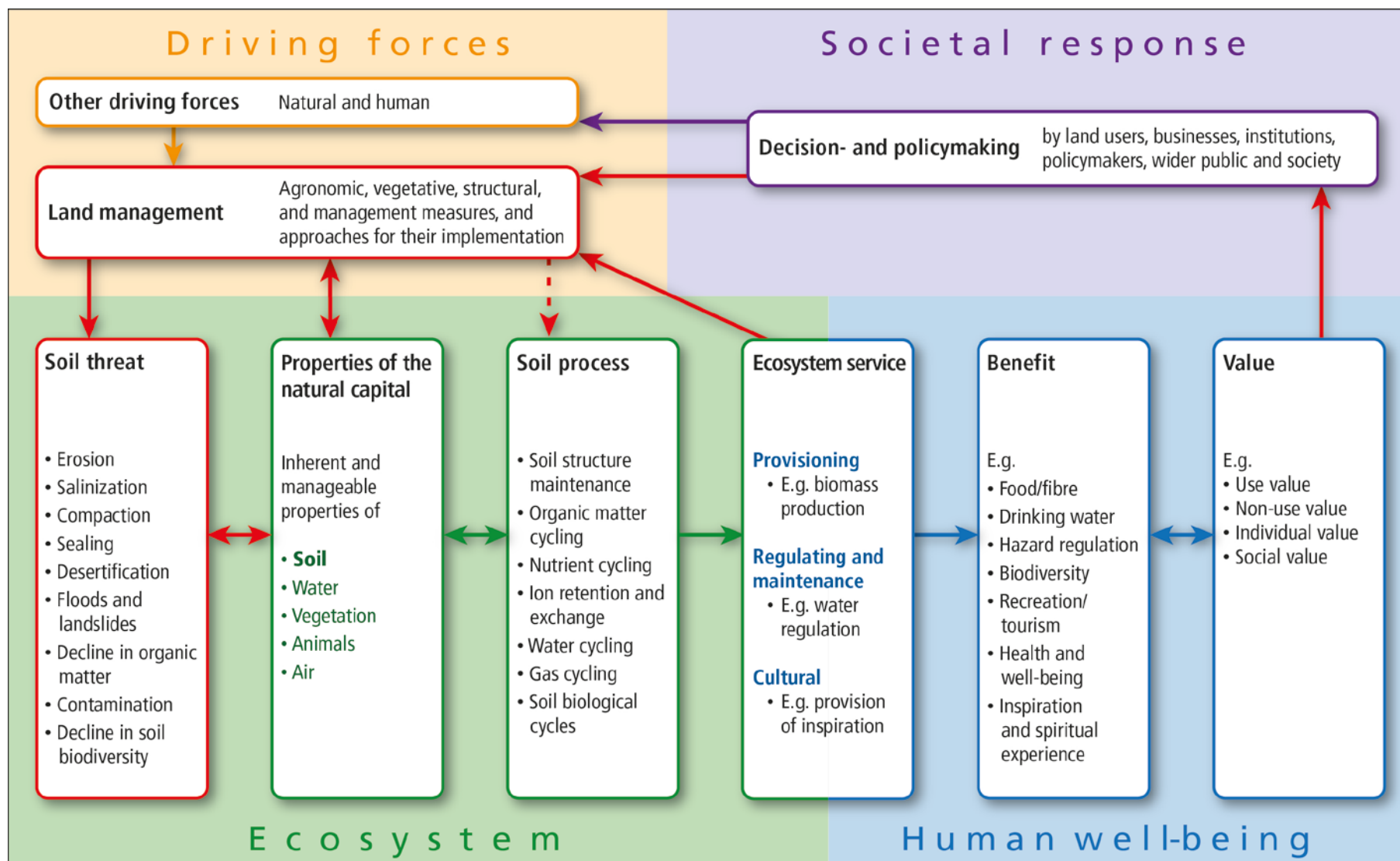


Figure 3: Proposed ecosystem services framework for RECARE. A detailed explanation is given in Sections 7 and 8.