



Convention on
Biological Diversity



Spatial units, scaling and aggregation

DRAFT

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¹*The views and opinions expressed in this report are those of the author and do not necessarily reflect the official policy or position of the United Nations or the Government of Norway.*

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0. Introduction

1. This report has been prepared as part of a project on Advancing Natural Capital Accounting² through testing of the System of Environmental-Economic Accounting (SEEA) Experimental Ecosystem Accounting. The objective of this report is to review the emerging concepts for spatial units, scaling and aggregation. It does so in the context of the SEEA Experimental Ecosystem Accounting (SEEA-EEA) (European Commission, OECD et al. 2013).

1. Links to SEEA Central Framework and SEEA-EEA

1.1 Discussion on links to EEA and how this guidance material is dealing with a particular issue

2. The SEEA-EEA presents a broad, coherent measurement framework for linking ecosystem extent, condition, capacity, services and values. Much knowledge and data exist individually on each of these topics. However, bringing it into an accounting framework both (a) assures consistency in concepts and classifications and (b) provides links to economic accounting.
3. With this in mind, the SEEA-EEA provides some initial principles and concepts with respect to spatial units, scaling and aggregation:

Spatial units are the basic building blocks for the analysis of location-specific attributes.

Scaling is the process of attributing information from one spatial, thematic or temporal scale to another.

Aggregation is the process of reducing many measures to simpler ones.

- **Spatial units** are the basic building blocks for the analysis of location-specific attributes. The SEEA Central Framework operates largely at the national level. The SEEA-EEA recommends a much finer spatial scale to compile information about ecosystems. For example, land cover change may be summarized at the national level. However, a land cover change matrix requires smaller spatial units to calculate what types of land cover changed and what they changed into.
 - **Scaling** is the process of attributing information from one spatial, thematic or temporal scale to another. Information on ecosystems, including their condition, services and beneficiaries occur on many different scales. Therefore, compiling ecosystem accounts requires guidance on how to attribute this information from one scale to another. This also includes the methods of transferring information from one location to another.
 - **Aggregation** is one aspect of scaling. It is the process of reducing many measures to simpler ones. When these measures are the same (such as dollars in the SNA), the process is relatively straightforward. When measures, units and scales are different, other approaches such as conversion to common units and the creation of indices are required.
4. Taking the SEEA-EEA as the point of departure, this report reviews recent literature and country experiences to (a) provide an overview of approaches used and (b) suggest means of further detailing the SEEA-EEA concepts and, perhaps expanding them to be more generally applicable.
 5. This report presumes the reader has a working knowledge of the SEEA-EEA. Training modules have been prepared as part of this project.

² See http://unstats.un.org/unsd/envaccounting/eea_project/default.asp.

1.2 Why is this important?

6. For any research-oriented multi-disciplinary initiative, it is essential to establish a common sense of existing concepts, measures, data and tools, but also to track the emerging ones. The SEEA-EEA research agenda (p. 155) includes the following objectives related to the purpose of this report:
 - *Delineating spatial units following the broad conceptual model outlined in SEEA Experimental Ecosystem Accounting. This should initially focus on spatial units for terrestrial areas (including rivers, lakes and other inland waters) and extend to units for marine areas and the atmosphere.*
 - *Investigating techniques for linking data related to ecosystem measurement to geo-referenced social and economic data. This multi-dimensional geo-referencing may be considered in the delineation of spatial units for ecosystems.*
 - *Examining aggregation methods for both ecosystem services and ecosystem condition indicators, to derive measures across and within ecosystems. In conjunction, methods of downscaling and upscaling information should be investigated.*
7. This report will identify some opportunities for advancing these objectives through further testing of the SEEA-EEA.

1.3 What is the issue being addressed?

8. This report addresses measures of spatial units, scaling and aggregation methods and approaches from an accounting perspective. It begins with a review of how these issues are represented in the SEEA-EEA and suggests how some of the areas of incompleteness may be informed by emerging work in the scientific literature and related ecosystem accounting initiatives.

2. Scope

2.1 What is in and why?

9. This report reviews each of the three main topics individually with appropriate links between them. For each topic, the current SEEA-EEA guidance is reviewed and specific issues are discussed. It then reviews how this topic has been addressed in ecosystem accounting and related research. Finally, it recommends some priorities for resolving these issues through further testing of the SEEA-EEA.

2.2 What is out and why?

10. For the purposes of this report, the spatial unit is discussed in terms of observable surface characteristics. This report does not address additional details of ecosystem classifications in detail. For example, several ecosystem classifications are based not only on surface characteristics, but also consider ecosystems to exist at different elevations (e.g., mountain) and depths (e.g., benthic coastal) and latitudes (e.g., tropical forest versus temperate forest). Such issues in ecosystem classification are reviewed in a separate report (Land Cover Accounting).
11. Existing models and other tools for analysing ecosystem services have their own spatial units, scaling approaches and methods of aggregation. These are not reviewed in this report, since (a) the amenability of these models to official statistics is not clear and (b) the details of the models are often not open to investigation. This would be a useful avenue for further investigation.
12. Although monetary valuation is one approach to aggregation, the topic is not treated in detail in this report. Valuation is the subject of another report in this series.

3. Discussion

3.1 Spatial units

13. An accounting framework requires statistical units about which information is collected and for which statistics are derived. This is much like economic and social statistical units in which the main categories of entities (government, business and households) are each further divided into types. A four-part hierarchy of location, establishment, company and enterprise, for example represents business entities in Canada. Each level in the hierarchy is associated with specific economic information that is available at that level (Statistics Canada 2012). For example, business locations are able to provide information on number of employees.
14. The SEEA-EEA suggests that the main statistical unit be spatially oriented. That is, since the objective is to compile information about ecosystems, the core statistical unit is a spatial unit that is seen as best representing measures associated with terrestrial (including open wetlands and inland water bodies) marine and coastal ecosystems.

*Since the objective of the SEEA-EEA is to compile information about ecosystems, the core statistical unit is a **spatial unit** for which measures associated with terrestrial (including open wetlands and inland water bodies), freshwater and marine and coastal ecosystems are compiled.*

SEEA-EEA representation

15. The SEEA-EEA recommends a hierarchical classification of spatial units, based on surface characteristics:
- **The Basic Spatial Unit (BSU)** is the smallest spatial area. That is, it is normally not further subdivided. It can be a remote sensing “pixel”, a larger grid cell (e.g., 1 km²) or a land parcel (such as represented by cadastral or ownership information).
 - **The Land Cover Ecosystem Functional Unit (LCEU)** is an aggregation of contiguous BSUs with homogenous characteristics (such as land cover, elevation, drainage area and soil type). An LCEU is classified into one of the 16 classes (**Figure 1**) in the provisional land cover classification. Many of the tables in the SEEA-EEA are based on aggregating other characteristics (such as extent, condition, service flows) over LCEUs of similar class. While not strictly delineating an ecosystem, the LCEU can be considered an operational definition for the purposes of ecosystem accounting.
 - **The Ecosystem Accounting Unit (EAU)** is a reporting aggregate of LCEUs. This may be a natural

Figure 1: SEEA-EEA Provisional land cover classification

- Urban and associated developed areas
- Medium to large fields rainfed herbaceous cropland
- Medium to large fields irrigated herbaceous cropland
- Permanent crops, agriculture plantations
- Agriculture associations and mosaics
- Pastures and natural grassland
- Forest tree cover
- Shrubland, bushland, heathland
- Sparsely vegetated areas
- Natural vegetation associations and mosaics
- Barren land
- Permanent snow and glaciers
- Open wetlands
- Inland water bodies
- Coastal water bodies
- Sea

unit, such as a drainage area, or an administrative unit, such as province, resource management area or state. The delineation of the EAU is relative to the reporting purpose, but given the hierarchical nature of the classification, LCEUs should not cross EAU boundaries.

16. Basic guidelines for compiling spatial units are provided in Annex 1.

Issues with spatial units

17. While the SEEA-EEA representation of spatial units is a useful starting point and may be considered a minimal set of criteria for spatial units, there are several issues, which deserve further discussion. These can be grouped into issues of delineation, choice of BSU and analytical limitations.

Delineation of spatial units

18. In terms of delineation, the current cover classification (**Figure 1**) applies best to terrestrial areas with only one level of vegetation canopy:
 - Being based on land cover, the treatment of freshwater, marine and sub-soil ecosystems is not well defined:
 - Freshwater ecosystems, such as streams, lakes, rivers and wetlands are often analysed as networks, with upstream and downstream areas having strong linkages.
 - Marine ecosystems, large rivers and lakes are also analysed in terms of depths. That is, seagrass beds and coral reefs exist below the surface. Benthic environments often constitute separate ecosystems from the pelagic ones above them.
 - Terrestrial sub-surface ecosystems, such as soil and caves, also exist below the surface, and are therefore not represented.
 - Dependence on satellite imagery alone will obfuscate certain important surface characteristics. For example, a canopy of trees may hide wetlands. Furthermore, small but important features, especially streams and wetlands may be smaller than one “pixel” in a satellite image.
 - Furthermore, an LCEU classification based only on land cover would ignore the fact that different parts of an LCEU may be under different management regimes. For example, an area of homogenous Forest tree cover may be in part protected and in part allocated to timber production. The implications for ecosystem services produced by these two sub-areas are quite different.
 - Similarly, different parts of an LCEU may exhibit different conditions, for example, levels of degradation. Assigning one set of average conditions to an entire LCEU would obscure this heterogeneity.
 - The treatment of airsheds and other connective phenomena are not defined in the SEEA-EEA:
 - Airsheds are similar in concept to watersheds in that they define areas affected by the same flow of air. They can be seen as airborne pathways of pollutants, pollen, seeds and leaves (Schindler 2009). Although airsheds could be an important element in understanding certain connections between LCEUs, it is not clear that they are coherently defined at the national level. For British Columbia, Canada, for example, airsheds are defined for only parts of the province where dispersal and concentration of pollution is defined as a management issue (Government of British Columbia, B.C. Air Quality 2014). They are often associated with urban pollution and may be defined to assess the dispersal of pollutants from a city to its surroundings.

- Stream flow and airflow represent two types of functional connections between spatial units. Another is the movement of animals between units that may be nearby or distant. Migratory birds may breed in one LCEU and spend their summers in another, thousands of kilometres distant. Pollinating bees maintain hives in the forest and emerge to forage in the fields during the daytime.

19. Therefore, the delineation of spatial units should be based on more than land cover information from satellite imagery. Additional information from other sources, such as hydrology, road networks and soil surveys can improve the creation of homogenous LCEUs. The delineation of atmospheric spatial units for ecosystem accounting is out of scope for this report, but could be informed by the above discussion on airsheds.

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Choice of BSU

20. The choice of BSU will impose different assumptions and approximations on the results and thereby affect the interpretation of the outcome. For example:

- Large (e.g., 1 km²) grids as BSUs, may include several land cover types and require either a decision as to the dominant land cover (e.g., if 40% of the area is Sparsely vegetated area, then the entire grid is classified as Sparsely vegetated area) or a statistical summary of the different land covers (e.g., 30% Forest tree cover, 40% Sparsely vegetated area and 30% Pastures and natural grassland)³. The former may be an unnecessary approximation if more detailed information is available. The latter is difficult to analyse statistically, since it is not known where within the cell each type exists.
 - These issues may not constitute major problems for large-scale analysis, such as at the national or global level, but since the intent of an ecosystem account is to integrate information at various spatial scales, using too coarse a resolution may impose unnecessary simplifications in this integration. An analogy in economic statistics is that a company may engage in multiple activities. The solution is to break up the company into analytical units (e.g., a strategic business units), which can be an aggregate of establishments conducting similar types of activities.
 - This may be related to the “ecological fallacy” (Schwartz 1994) described in sociological analysis where causal inferences about individual behaviours are made from group data. In the case of spatial units, large heterogeneous units can be considered “group data” since they are an aggregation of more detailed data. As shown by Kallimanis and Koutsias (2013), inferences about the land cover diversity at the country level differ depending on whether they are based on small-scale or larger-scale aggregated data. This is further discussed in the next section on **Scale and Scaling (Issues in scale and scaling)**.
- Using a cadastre⁴ as a BSU, while useful for distinguishing the management regime of a plot of land, assumes that the plot is homogeneous in terms of surface properties. That is,

³ As in business statistics, these are known as “complex units”.

⁴ A cadastre is a register of property showing the extent, value, and ownership of land for taxation.

the same issues as the large grid apply; using this information alone risks oversimplification.

21. Therefore, spatial information should be maintained in an appropriate level of detail. If the purpose of the ecosystem account is to generate only national-level aggregates, then large (e.g., 1km or larger) BSUs may be appropriate. However, if the purpose of the ecosystem account is also to “drill down” to investigate local phenomena, smaller-scale (e.g., 30m) BSUs may be required.

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Analytical limitations

22. Are LCEUs necessary? Some implementations of ecosystem accounting (Eigenraam, Chua et al. 2013, Sumarga, Hein 2014) attribute all information to the BSU level and then generate analyses for larger-scale areas (e.g., drainage areas, conservation areas, ecosystem types) as required. This may avoid one set of issues, such as delineating homogenous LCEUs and aggregating conditions or services to the LCEUs. However, for more comprehensive ecosystem accounts, this approach may complicate compilation and analysis. As with business units, certain information, such as land cover, is most amenable to smaller units. Other information, such as proximity to human habitation and indices of species richness, might best be calculated once for a larger area, such as an LCEU and analysis done at that level. Furthermore, some aspects of ecosystem condition, such as fragmentation⁵, will require consolidation at higher levels of spatial aggregation than the LCEU.
23. There is a concern about the delineation of LCEUs over time. Whereas BSUs and EAUs are relatively time-invariant, LCEUs are not. That is, if a BSU represents a homogenous grid cell and if its properties change over time, changes over time can be represented in terms of how that BSU has changed. If an EAU represents an administrative (e.g., conservation area) or natural unit (such as drainage area or ecozone), its boundaries are unlikely to change significantly over accounting periods. However, LCEUs represent aggregates of BSUs, some of which may have changed over the accounting period. Using the same LCEU for two accounting periods would introduce additional approximations. For example, classifying an LCEU by the majority of land cover in the BSUs it incorporates would downplay the influence of small changes over time. Defining new LCEUs for each accounting period would remove this source of error.
24. As with business statistics, information about ecosystems exists at various scales (e.g., point data from local field studies; spatial areas, such as conservation areas and species ranges; and networks such as roads, streams and ecological corridors). Attributing information from larger scales to smaller ones (or downscaling, see next section on **Scale and Scaling**) assumes uniformity within the larger scale. For example, information about crop productivity (in tonnes/ha) may be available at the farm level, or higher aggregates such as Census Agglomerations in Canada. Allocating the same productivity value to each BSU within the farm, which may span hundreds of hectares, assumes that the biophysical conditions (such as slope and soil type) throughout the farm are uniform. Aggregating this value to LCEUs, based on these biophysical conditions runs the risk of introducing an unnecessary source of error. That is, LCEUs with different average biophysical conditions would be attributed with the same aggregate level of productivity. Analysis at the BSU level, such as correlating average productivity with the more specific biophysical characteristics

⁵ Fragmentation is a measure of the degree to which an ecosystem is divided into smaller areas by human built infrastructures such as dams, roads, railways, pipelines and electrical infrastructure.

would downplay that relationship. However, if information on crop productivity were allocated to a level higher than the LCEU (e.g., drainage area), average (or dominant) biophysical characteristics would likely correlate better with average productivity.

25. The above example suggests that there are benefits to maintaining a richer set of spatial units than simply BSU, LCEU and EAU. Rather than scaling all data to the BSU or LCEU level, maintaining data (for example, on conservation areas, administrative data, species ranges, soil classes and drainage areas) would ensure that biases introduced by scaling are minimized.
26. Any approach to spatial units will impose certain analytical limitations:

- Too rigid an approach will limit the ability to integrate information from various spatial scales. Furthermore, requiring certain data at certain levels will be difficult to implement since those data are not always available in all countries (for all periods).
- Too flexible an approach will leave many choices and calculations to be made at the analysis stage. For example, having overlapping spatial units would require scaling and allocation for each analysis. While this is feasible if all analysis were to be done using GIS, not all participants in an ecosystem accounting team will necessarily be using GIS.
- What size of spatial unit is most useful? This will largely depend on the analytical objective of the account. Larger BSUs will be sufficient for analysis at the national level and for assessing general trends. However, the power of ecosystem accounting is the capacity to “drill-down”, spatially and thematically to be able to address more specific issues. For example, if wetlands are found to be changing in extent or condition at the national level, it would be useful to use the same spatial database to assess where changes are occurring, what is causing the changes and the implications of those changes for local and national well-being.

There are benefits to maintaining a richer set of spatial units than simply BSU, LCEU and EAU. Rather than scaling all data to the BSU or LCEU level, maintaining data at their appropriate scales would ensure that biases introduced by scaling are minimized.

Examples of implementation

27. While not all the issues discussed above are addressed in the literature, there are several examples of national experiences that could point the way to further testing.

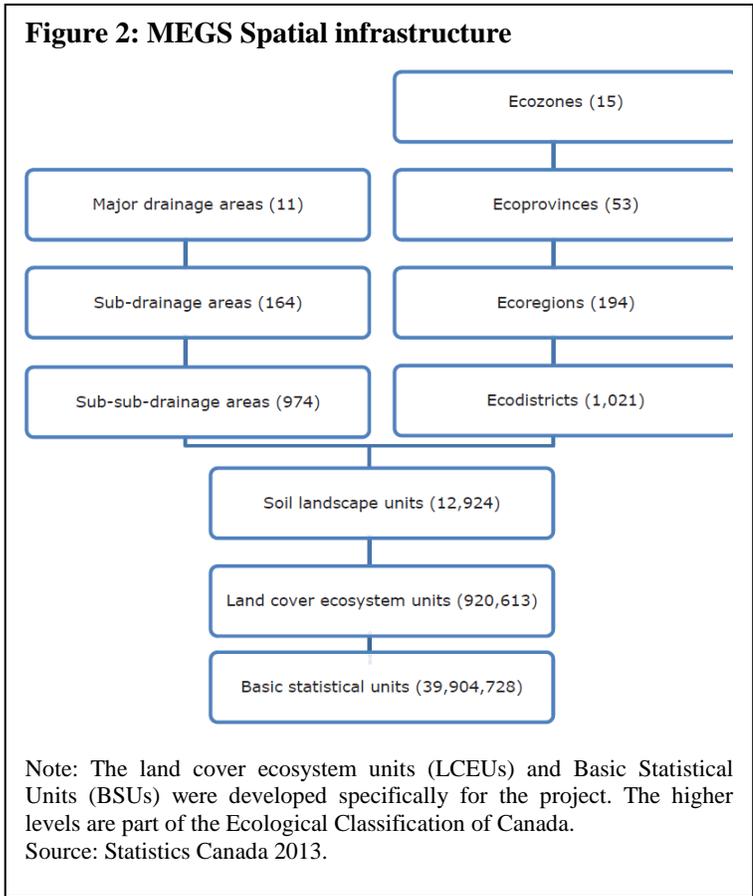
Canada’s Measuring Ecosystem Goods and Services (MEGS) project

28. The previous section discussed several issues related to LCEU delineation. As Canada has applied the concept of LCEUs in its program, this experience can be drawn upon to address these issues. The Government of Canada’s Measuring Ecosystem Goods and Services project (MEGS) (Statistics Canada 2013) applied a stricter definition of LCEU than the core criteria defined in the SEEA-EEA. Firstly, the spatial classification framework was placed within the existing national hydrological and ecological classifications (**Figure 2**). Secondly, the delineation, while based on 250m MODIS land cover data, was augmented by more detailed hydrological data (to better distinguish streams and wetlands), data from an analysis of Census blocks (to better define settled areas), a detailed road network file and digital elevation data (**Figure 3**). This allowed for the more detailed delineation of LCEUs.

29. An LCEU was defined in MEGS as an area of homogenous land cover, elevation, slope, soil type and “ruggedness” that did not cross a major road, rail line, electrical transmission line, watershed (height of land) or stream. In all, from a core of over 39 million BSUs, 920 thousand LCEUs were defined. Cadastral information was not used, since this is not available at the national level for Canada.

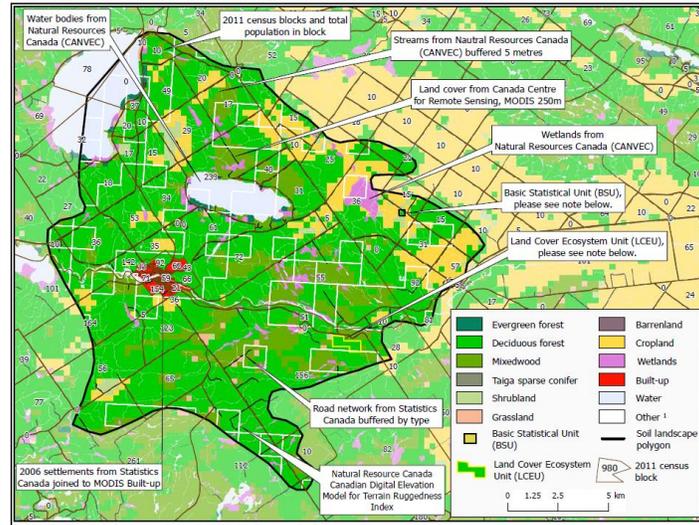
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30. Data for LCEUs (landscape type, average natural parcel size, average distance to natural land parcel, barrier density, wetland extent, peatland extent, population density, land in agriculture, livestock density, streamflow variability, land area fertilized, nitrogen manure from livestock, phosphorous in manure from livestock, were aggregated to the sub-drainage level and reported for all 164 sub-drainage areas in Canada.



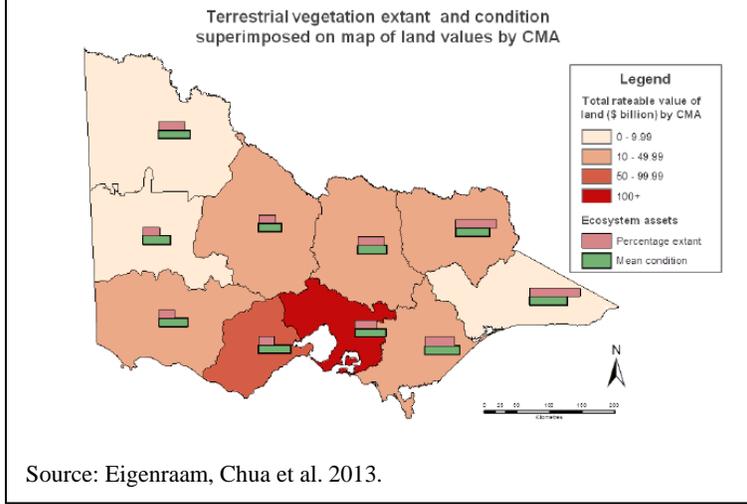
31. MEGS also developed a classification of coastal and marine areas (based on marine ecosystems) and rivers (based on defining areas of upper, middle and lower drainage area), but this is not yet published. For the 2013 publication, marine areas used for marine and coastal biomass extraction were based on Fisheries and Oceans Canada’s statistical areas.

Figure 3: MEGS Example of delineating LCEUs



Source: Statistics Canada 2013.

Figure 4: Victoria terrestrial vegetation and land values



Source: Eigenraam, Chua et al. 2013.

The Government of Victoria's ecosystem accounts

32. The Government of Victoria's (Australia) ecosystem accounts (Eigenraam, Chua et al. 2013) based most of its land asset account on 100m resolution satellite imagery. The EnSym⁶ tool used for the spatial analysis is raster-based, therefore the analytical tables and maps (See **Figure 4**) were generated by aggregating BSU characteristics (vegetation type, land use, tenure, protection status, mean condition, landscape context) to Catchment Management Authority (CMA), statistical areas (SA1) and bioregions for the province.

⁶ <https://ensym.dse.vic.gov.au/cms/>

Australia's Land Account: Queensland, Experimental Estimates

33. Australia's Land Account: Queensland, Experimental Estimates, 2013 (Australian Bureau of Statistics 2013) were based on cadastral boundaries. This provided information on both land value and land use. The cadastre spatial data was intersected with 250m resolution land cover data, creating spatial units that were sometimes smaller than the original grid. They cautioned the interpretation of any areas of less than 6.25ha in area (250m x 250m), since the land cover grid was already an estimate of the predominant land cover in that grid (see choice of BSU above). That is, if a cadastral unit intersected with a portion of a land cover grid, there would be no certainty that the land cover for that portion was actually represented by the dominant land cover of the 250m grid.
34. Summary statistics were reported at the NRM (Natural Resource Management) region level (15 of which exist in Queensland) and SA1 level (Statistical Area 1, 11,039 of which exist in Queensland). The Statistical Areas had been previously created for Census purposes and represent an average population of about 400. Since cadastral boundaries did not align with SA1 boundaries, cadastral plots were assigned to SA1 areas based on the location of their geographic centroid.

Sumarga and Hein

35. Sumarga & Hein (2014) as noted earlier, attribute information for Central Kalimantan to the BSU level. Maps of seven ecosystem services (timber production, rattan production, oil palm production, paddy rice production, carbon storage, carbon sequestration, orangutan habitat suitability) from various sources including land cover, soil, digital elevation (90m resolution). Topographic and hydrological maps were converted to a raster format at 100m resolution. The combined information was used to develop models of potential suitability for these services, based on land characteristics.

The SCBD Quick Start Package

36. The Secretariat of the Convention on Biological Diversity Quick Start Package (Weber 2014) recommends a data model in which all data are assimilated to the same grid level (**Figure 5**). In many cases, this would be a 1 km² grid, but finer resolutions would be feasible. Weber suggests a variety of analytical and reporting spatial units including:
 - Land Cover Ecosystem Functional Units (LCEUs): defined as in SSEEA-EEA but also contain linear features such as rivers and ecotones (ecosystem boundaries);
 - Socio-Ecological Landscape Units (SELU): The definition and use of SELUs are not easily distinguished from LCEUs. The Socio-ecological system is defined as essentially the ecosystem plus the remotely-sensed land use (largely agriculture and urban) of the ecosystem.
 - Marine Coastal Units (MCU): Based on sea-bottom (benthic) vegetation and physical characteristics;
 - River System Unit (RSU): Comprised of homogenous stream reaches (HRSUs) of the hydrological network (**Figure 6**)
 - Homogenous stream reach units (HSRUs): Are segments between two confluences (For example XD1 in **Figure 6**).

Figure 6 River basins, rivers and HRSUs (Weber, 2014)

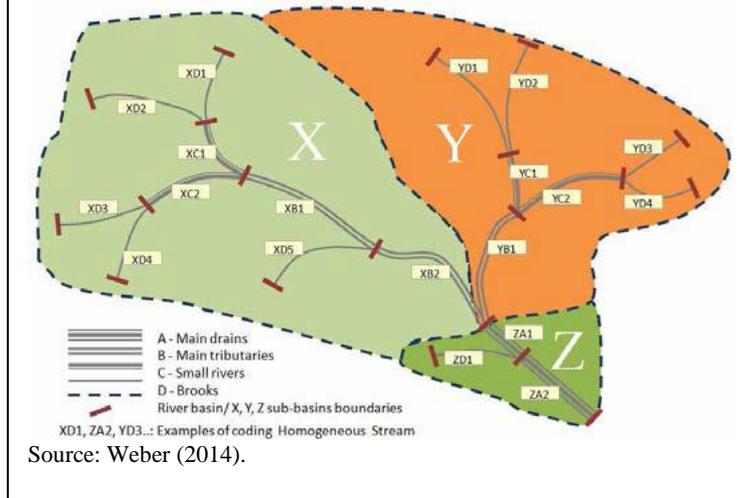
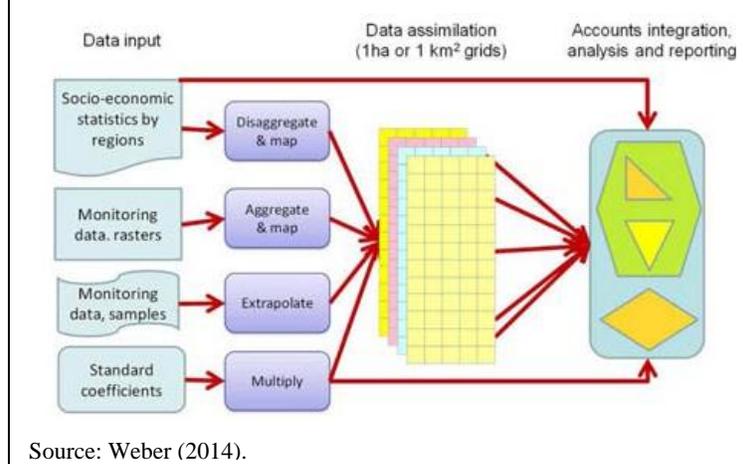


Figure 5 Suggested data framework (Weber, 2014)



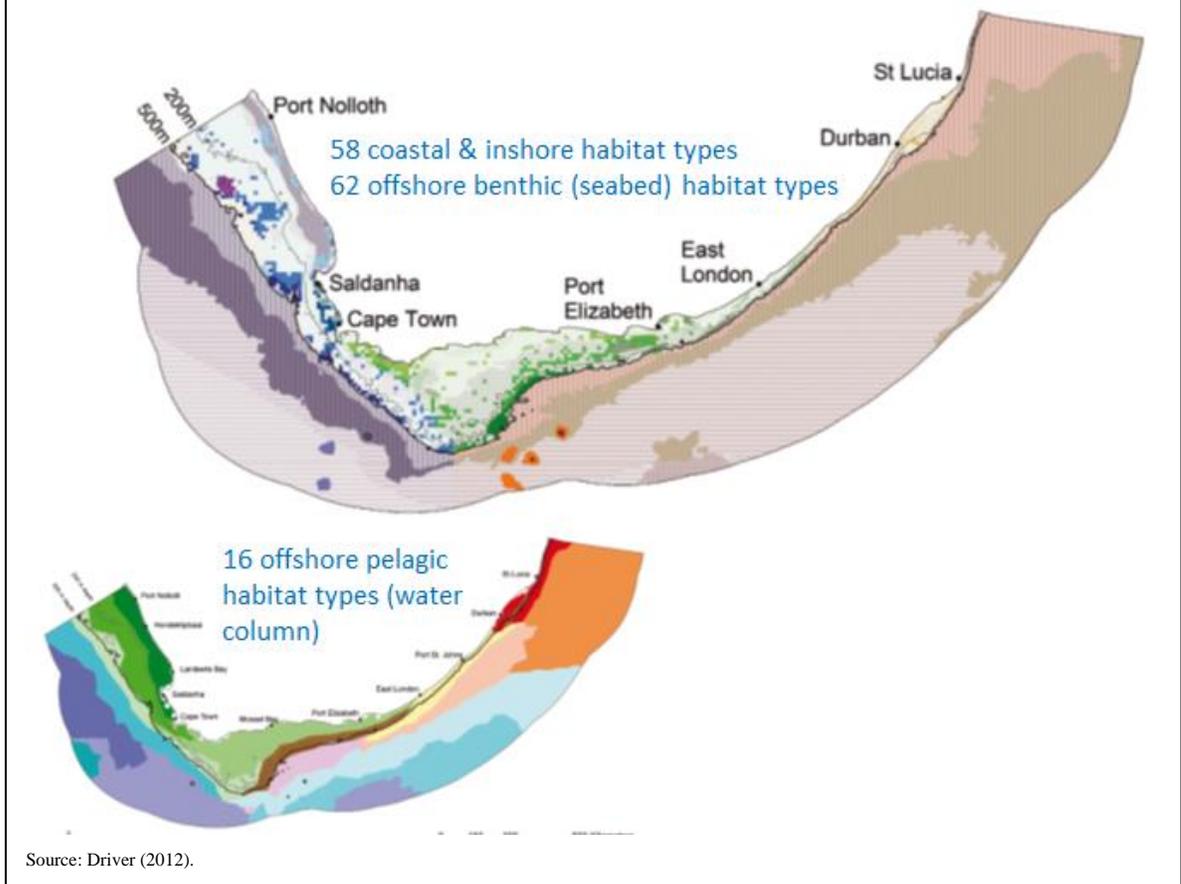
European Environmental Agency Experimental Ecosystem Accounts

37. The experimental ecosystem accounts produced by the European Environmental Agency present a broad picture of more than 30 countries within a standard grid of 1 km². Aggregation is done by (a) socio-ecological landscape units (SELU), which are defined as a combination of land cover and land use, and (b) river sub-basins (about 600 units for all of Europe) (Weber 2011).

South Africa’s approach to delineating coastal, marine and river ecosystem types

38. SANBI, the South African National Biodiversity Institute (Driver 2012) has developed a classification of offshore pelagic and benthic coastal and marine habitat types (**Figure 7**). Factors used to classify coastal and inshore habitat types included substrate (e.g. rocky, sandy, muddy, gravel, mixed), wave exposure (sheltered, exposed or very exposed), grain size (important for determining beach type), and biogeography (ecoregions). Factors used to classify offshore benthic habitats included depth and slope (shelf, shelf edge, deep sea), substrate (e.g. hard or unconsolidated soft and gravel habitats), geology (e.g. sandy, muddy, gravel, reef, hard grounds, canyons and ferro-manganese deposits), and biogeography. Factors used to classify offshore

Figure 7 Coastal, inshore and offshore habitat types for South Africa



pelagic ecosystems included sea surface temperature, primary productivity and chlorophyll content, depth, turbidity, frequency of eddies, and distribution of temperature and chlorophyll fronts.

39. While this last example will also inform the ANCA companion report on Land Cover Accounting, it also illustrates the use of biophysical information in the delineation of coherent marine and coastal spatial units. Further, it demonstrates the feasibility of distinguishing benthic from pelagic ecosystems.
40. SANBI implements a similar approach as that suggested by Weber (2014) in delineating river catchment segments into four slope categories mountain stream, upper foothill, lower foothill and lowland river. This has been used in combination with ecoregion and flow variability to identify 223 river ecosystem types.

Recommendations for further testing of spatial units

41. Given the multitude of approaches that have been used in delineating spatial units, the further testing of the SEEA-EEA could support the development of best practices for ecosystem accounting in terms of spatial units by:
 - Testing approaches to delineating LCEUs (beyond satellite imagery) including freshwater, coastal and marine units,
 - Developing criteria for and testing other intermediate spatial units (such as landscapes, viewscales and river units),

- Linking levels of spatial units with specific information,
- Testing the implications of treating LCEUs and other intermediate spatial units as time-invariant in comparison with redefining intermediate units for each accounting period, and
- Investigating how spatial units are treated in various ecosystem services models.

Further testing of the SEEA-EEA could develop best practices for:

- *Delineating homogenous LCEUs including freshwater, coastal and marine units*
 - *Delineating other intermediate spatial units (landscapes, viewscales, river units)*
 - *Linking spatial levels to specific information*
 - *Analysing changes in LCEUs over time*
- How spatial units are delineated and treated in spatial models should also be investigated.*

Testing approaches to delineating LCEUs

42. Various approaches have been used to delineate LCEUs, ranging from land cover only to more detailed ones including hydrographical, topographical and socio-economic information. Further testing of the applicability of these approaches will likely be limited by available data and analytical tools, but could result in a set of best practices within those limitations. The approaches suggested by Weber (2014) for delineating river units and by Driver (2012) for delineating coastal and marine units should be further tested. Results of such testing would contribute to improvements in ecosystem classification as well.

Developing criteria and testing other intermediate spatial units

43. Another avenue for future research would be to define a “landscape” more rigorously. Much research is done on landscapes but with little insight into the rigorous definition of the concept. For example, Raudsepp-Hearne et al. (2010) consider a landscape as a municipal administrative unit. DeGroot et al. (2010) suggest that landscapes are a unique ecological scale. That is, by the processes that occur between ecosystems (or in the case of ecosystem accounting, LCEUs) provide a layer of services that are better analysed and managed at that level. Landscapes are sometimes taken to be synonymous with viewscales, that is the land that is visible from a particular vantage point (Bin, Crawford et al. 2008).
44. In the operationalization of the concept of *ecosystem* for SEEA-EEA accounting purposes, the simplifying assumption was made that land cover provides a useful starting point for delineating surface areas with strong internal ecological relationships. For example, an area of forest tree cover will exhibit representative ecological relationships among plant species, fauna, microbes and the abiotic components. An area of open wetland will also exhibit different, but no less representative, ecological relationships among its components. If these two ecosystem types exist in proximity to one another, there are also relationships between the components of the two. That is, the vegetation in the forest tree cover area may affect the water flow and the amount of detritus going into the wetland. The forest tree cover area may also provide a nesting ground for the bees and birds that feed in the wetland. Conversely, the wetland may provide a source of flood control, nutrients and water supply for the adjacent forest area.
45. Although they are described conceptually in the SEEA-EEA, such inter-ecosystem flows are not operationalized in the current document in terms of accounts for Assets, Condition, Production, Biodiversity and Supply-Use.
46. Weber’s (2014) description of a Green Ecotones Index may provide a starting point for a more rigorous definition of landscape. The index is a measure of the importance of interfaces between ecosystem types. Extending this concept, it should be possible, using only LCEUs and some of

their known biophysical properties (such as land cover, land use, elevation, slope, catchment position), many of which could be remotely sensed, to define clusters of LCEUs with strong intra-LCEU relationships. This spatial approach could augment the “*conceptual linkage between CICES, ecosystem type, function and intermediate services*” suggested in an accompanying report (Bordt 2015). That is, by delineating clusters of LCEUs in terms of the strength of the relationships between them, the need for separate treatment of inter-ecosystem flows would be addressed.

47. In addition, this approach could include socio-economic attributes. For example, parks near residential areas could be considered to have a strong socio-ecological affinity. Furthermore, it could be combined with Luck et al.’s (2009) concept of the Service Providing Unit-Ecosystem Service Providers (SPU-ESP) continuum. That is, codifying these associations between habitats would inform the measurement of attributes to link ecosystem condition with capacity. This is discussed in more detail in an accompanying report (Bordt 2015).

Spatial scale	Data	Type of analysis
BSU	Land cover, location	Land cover change
LCEU	Land use, soil type, slope, elevation, location within catchment, species abundance, biomass	Local service production, local service-beneficiary linkages
Landscape	Barriers, habitats, ecological interactions, beneficiaries, micro-climate, local drivers of change (e.g., population, industry), visitor rates, streamflow, erosion rates	Fragmentation, heterogeneity, inter-ecosystem flows, biodiversity
Drainage area	Freshwater availability, recharge rates	Water-based phenomena such as flow of water, pollutants and nutrients.
EAU	Management regime, environmental activities (expenditures, management), beneficiaries	Aggregate of all of the above.
National	Socio-economic drivers, beneficiaries	Trends in all of the above; national beneficiaries
Global	Climate, socio-economic drivers, beneficiaries	Global trends in all of the above; global beneficiaries;

Linking levels of spatial units with specific information

48. Developing intermediate spatial units would be informed by insights into not only which information was appropriate to which level, but also which types of analyses were appropriate for each level. **Table 1** provides a starting point for discussion. This could be linked to the discussion of institutional and policy scales in as discussed in an accompanying report (Bordt 2015).

Testing the delineation of LCEUs over time

49. Testing the delineation of LCEUs over time would provide insights into (a) the effort required to re-define LCEUs for each accounting period, (b) the availability of the necessary data to define them and (c) the statistical and accounting implications of the two approaches.

Investigating how spatial units are treated in various ecosystem services models

50. The ecosystem services models investigated as part of this project, as discussed in an accompanying report (Bordt 2015) should be further investigated in terms of their treatment of spatial units. This could be made part of a larger review of the properties (scaling, aggregation, indicators, treatment of error, etc.) of these models.

Amenability to official statistics

51. National Statistical Offices (NSOs) may already possess much of the data and many of the tools required to participate in the testing of spatial units:
- **Standardization of spatial units:** For example, Statistics Canada, in collaboration with other departments has developed a [Standard Drainage Areas Classification](#), which serves as a common point of departure for aggregating socio-economic information to this spatial unit.
 - **Provide data:** The NSOs may possess standardized spatial data on infrastructure (roads, railways, electrical infrastructure) and socio-economic characteristics (population, dwellings, income) by small-areas.
 - **Integrate spatial data in GIS and dissemination:** The NSOs may have experience in integrating disparate spatial datasets and disseminating spatial data via web services.
 - **Apply methods and approaches:** Several statistical methods will be required such as cluster analysis to test and construct landscapes.

3.2 Scale and Scaling

52. The previous section on spatial units has already identified some issues with respect to spatial scale. This section focusses on the process of scaling. That is attributing information from one scale to another. This applies not only to spatial scales, but also to thematic scales (e.g., classifications of land cover, LCEU types and species groupings) and temporal scales (daily, monthly, seasonal, annual, etc.). This section also provides a summary and review of issues in transferring information from one location to another (e.g., benefits transfer), which shares many conceptual linkages with scaling.
- All aspects of ecosystems, their functions, processes, conditions, services and beneficiaries occur at various spatial and time scales. Furthermore, simple or more complex classifications are used to analyse these. Ecosystem accounting needs to be aware of these scales and develop appropriate approaches to scaling from one to another.*
53. Scaling from a general business or national accounting perspective, is usually relatively straightforward. Many economic measures can be scaled by simple adding or averaging them. For example, monthly income for 12 months adds to annual income. The monthly number of employees can be averaged to represent annual employment. Both can be simply summed to create industry or regional totals. Establishments are classified into industry sectors based on simple rules such as the nature of the main revenue producing line of business. Quarterly GDP can be estimated by adjusting the previous year's GDP by more recent monthly information. Uncertainty in the SNA is usually expressed in terms of the size of the revisions. Annual GDP estimates require revisions since they are often based on business sample surveys, whereas tax data for benchmarking may only be available one or two years after the reference year.
54. Accounting for ecosystems is more complex: the metrics themselves are not necessarily amenable to simple scaling approaches. Furthermore, there is less agreement on what to measure, how to classify it and how to treat the data to represent ecological complexity over a range of scales.
55. Components of ecosystem accounts occur at various spatial and temporal scales. Drivers of change may be global and slow, such as climate change, or they may be local and fast, such as land-use change near a wetland. Similarly, ecological processes can occur over a range of spatial and temporal scales: species may migrate seasonally between hemispheres and an individual tundra plant may absorb nutrients from its local surroundings during the daytime over a short growing season.

56. Ecosystem services also are generated and used at a broad range of scales: carbon sequestration, generated locally, provides benefits at a global scale whereas subsistence farming is a local service providing benefits for local beneficiaries. As further pointed out by Hein *et al.* (2006), stakeholders and institutions also occur at various spatial scales. As Hicks (2011) also suggests, considering multiple scales, a broad range of ecosystem services and decision contexts is essential to understanding the benefits of ecosystems to human well-being.
57. The issue of spatial scales relates back to the topic of **Spatial Units** in that, given the operational definition of *ecosystem*, wherein interactions within the ecosystem are stronger than interactions with elements outside the ecosystem. For some analytical purposes, an ecosystem could consist of all plant, animal and micro-organism communities in the world. They could also be overlapping or nested. “Consequently, for accounting purposes, **ecosystem assets** are defined through the delineation of specific and mutually exclusive spatial areas” (SEEA-EEA p. 162).
58. Furthermore, information about the characteristics of ecosystems may be collected at various spatial, temporal and thematic scales. Specific studies may relate to point locations (e.g., water quality monitoring stations), plots (e.g., species abundance, biomass production), landscapes (e.g., visitor rates, stream flow, erosion rates), drainage areas (e.g., freshwater availability, recharge rates), or administrative boundaries (e.g., expenditures on ecosystem rehabilitation and maintenance of protected areas). In terms of temporal scale, they may use, among others, daily measurements, an annual accounting period or 20-year age classes. In terms of thematic scale, studies may represent land cover in a few classes (as few as three or four) or many classes (as many as 40-50).
59. Given this heterogeneity, it is important to understand how and when to attribute information from one scale to another. That is, information is available at one scale (e.g., drainage area), yet it is required at a higher (e.g., national), lower (e.g., LCEU) or other (e.g., provincial) spatial scale. Using appropriate methods for scaling will ensure that the results are as statistically robust as possible.
60. It is also crucial to understand when information can be transferred from one location to another. For example, if the phosphorous absorption rate is measured for one wetland, can this value be used to represent the phosphorous absorption rate of another, similar wetland?

SEEA-EEA representation

61. The SEEA-EEA addresses scaling in terms of:
 - Spatial scaling (para 2.107): Downscaling, upscaling and aggregation of spatial data are discussed, but not treated in detail. There is also a general discussion of some spatial scaling issues such as the scale problem, boundary problem and modifiable area unit problem.
 - Transferring information (para 2.11): Also discussed, but not detailed are benefit transfer methods (value transfer, scaling up, benefit transfer, meta-analysis) used to apply data from one site to another or to multiple sites (also para 5.124).
62. The SEEA-EEA does not specifically discuss issues in temporal or thematic scaling. Furthermore, scaling of data, spatially, temporally or thematically, requires the appropriate treatment measurement of sources of error and variance (uncertainty).

Issues in scale and scaling

63. This sub-section will present a review of some issues in spatial, temporal and thematic scaling and their treatment. While any one of these topics would be deserving of a textbook, it is important in

any testing of the SEEA-EEA that non-statisticians, statisticians and accountants be aware of these issues and seek further expertise when required.

64. Most if not all statistical analysis involves the revelation of patterns in the underlying data. One objective of the SEEA-EEA is to establish a coherent framework for national-level interpretations of ecosystem assets, conditions and services, while maintaining a multi-scale capacity to support the integration and interpretation of local conditions.

Scaling of data in ecosystem accounting requires attention to:

- *The scale of the phenomenon being measured*
- *Sources of error and their treatment*
- *Underlying patterns in the data*
- *Uniformity of distribution of underlying data*
- *The amenability of the data to scaling*

65. The general issue here is that whenever data are transformed by downscaling, upscaling or classification into categories, assumptions are made about the distribution of those data. Furthermore, scaling and aggregation of data often implies an increase in uncertainty or a loss of information⁷. If the uncertainty outweighs the certainty, we run the risk of basing interpretations on spurious correlations. This leads to the situation where studies of the same phenomenon reach different conclusions. There are statistical approaches for treating the assumptions and reducing sources of uncertainty, but they tend to be hidden in the details of all but the most diligent scientific papers. Since there is little case history of what scales and which scaling approaches are appropriate for ecosystem accounting, issues of scale and scaling require further description and incorporation into SEEA-EEA testing.

66. This section will discuss these issues in scaling in terms of the scale of the phenomenon, sources of error, underlying patterns, uniformity of distribution, amenability to scaling and treatment of sources of error.

The scale of the phenomenon

67. The first question every analyst should ask is “**What is the scale of the phenomenon we are trying to measure and understand?**” The second question should be “**What are the sources of error and how can they be minimized?**”

68. Both questions can be illustrated by the simple problem of measuring the coastline of a country. This is not as trivial as it may seem. Tracing the coastline using maps made with 1km resolution will return an answer in kilometers. Using maps with a higher resolution, say 30m, will likely result in a longer coastline since there is more variation. At this scale, one would need to decide, for example, where an estuary stops becoming part of the coastline and starts becoming part of the river. At this scale, the coastline would also be susceptible to tidal variations, leading to complications of whether the map is representative of high tide, low tide or an average. At an even smaller resolution, the coastline would be measured as being even longer with more variation and variability. This *Coastline Problem* inspired Mandelbrot (1967) to develop the concept of fractals, that is, curves of fractional dimensions with lengths that diverge to infinity. The conclusion is that coastlines are infinite, but that measuring the change in variability at different scales provides a measure of that complexity.

⁷ |A few examples may illustrate the need for diligence in transforming data. When combining two sources of data, such as an index of two water quality measures, the uncertainty of the index may be additive or multiplicative. A more detailed source of spatial data may be used to reduce the uncertainty in a less detailed source of data, such as using hydrology data to delineate wetlands within remotely sensed land cover data. This presumes that the hydrology data are more precise than the land cover data. In creating classifications or “bins”, uncertainty is decreased since the resulting bins contain less information than the original source.

69. The simple advice on dealing with such issues is to choose appropriate spatial, temporal and thematic scale for the phenomenon that we are measuring and the scale of the decision to be made with it. In the above example, for comparing the coastlines of nations, a moderate resolution would likely provide a sufficient and relatively constant measurement over time. However, if coastline change of a small island were to be monitored over time, a higher resolution may be advisable. Furthermore, at this scale, temporal variability over seasons and tidal cycles would also need to be taken into account.
70. Cox and Moore (2010) provide an example of the scale of the study area for analysis of species associations. They note that as the size of a study area increases, so does the number of species recorded in the same area. This increases the appearance of positive correlations between species. Conversely, smaller study areas will record fewer species in a given area and thus show more negative associations. They relate this to the issue of selecting appropriate boundaries (discussed below) and whether or not the boundaries between ecosystems are sharp or diffuse.
71. The next set of questions involve the **upscaling** of data. That is, creating spatial, temporal or thematic aggregates. This is often necessary not only for simplifying the analysis, but for comparing results over time and between instances. The SEEA proposes standard classifications of land cover, land use and ecosystem services. As well, it proposes indicators of ecosystem condition and flows of services that are scaled to the accounting period and reported on spatial units (LCEU, EAU, country). Classifying data into the established categories and aggregating by spatial units and reporting periods all make approximations of the underlying data.

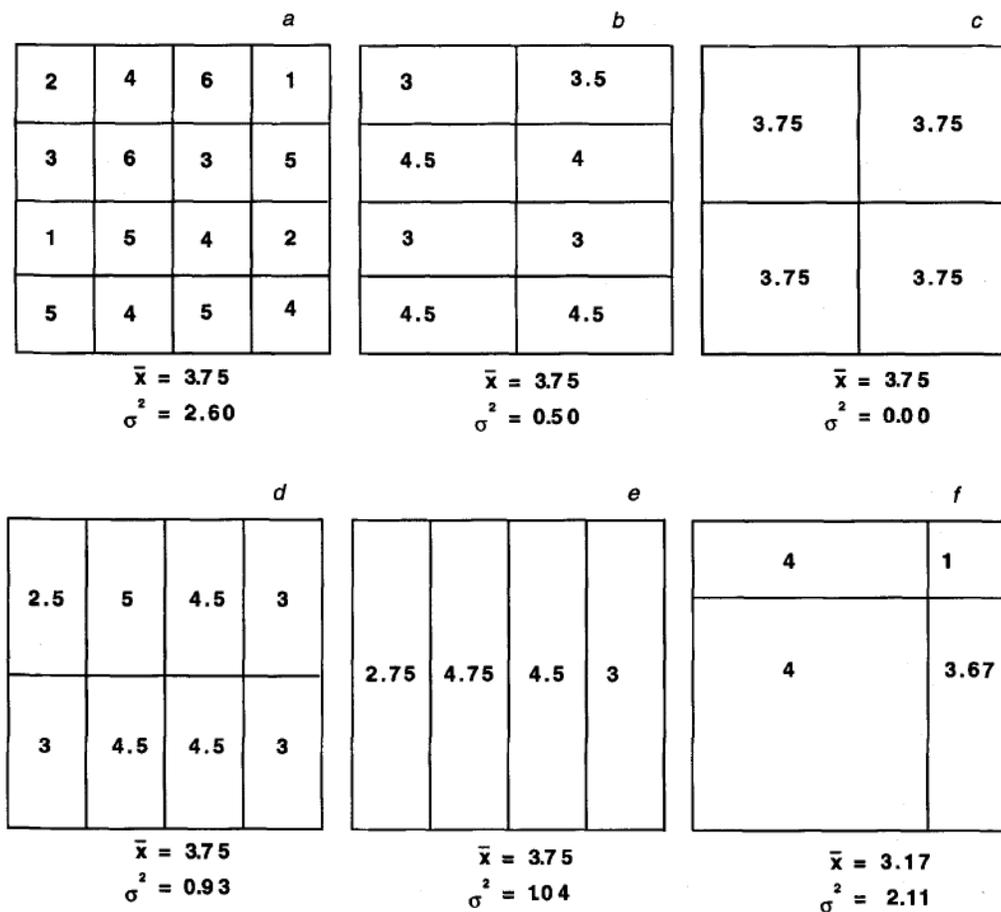
It is important to choose appropriate spatial, temporal and thematic scale for the phenomenon that we are measuring and the scale of the decision to be made with it.

Underlying patterns

72. The third question every analyst should ask in this context is “**Does my classification obscure or emphasize the underlying patterns in the data?**” The general advice here is to understand the distributions of the data before creating classes. For spatial analysis, this illustrates the **boundary problem (or modifiable area unit problem, MAUP)**. That is, by selecting arbitrary boundaries (such as administrative areas or cell grids) for the analysis of ecological spatial associations, this will downplay the influence of neighbours, gradients and spatial patterns. For this reason, analysts often choose ecological boundaries (ecozones) for the analysis of ecological relationships and drainage areas for the analysis of water-based relationships. In the SEEA-EEA, the delineation of the LCEU is one suggested approach to treating spatial areas that reduce this boundary problem. That is, land cover is one important determinant of the patterns (of conditions, interactions and services) that need to be represented. Whether it is the most important remains to be tested.
73. Jelinski and Wu (1996) illustrate the combined effect of aggregation and zoning (the selection of spatial units) in spatial analysis as shown in **Figure 8**. They suggest five alternative approaches to addressing the issue in landscape ecology:
- *A basic entity approach*: This refers to the selection of ecologically-significant entities (trees, individual animals, gopher mounds, patches) that are not subsequently modified or aggregated. However, they note that this approach does little to address issues of inter-relationships that require integration of spatially distributed data. Further, whether or not

Patterns in underlying data can be obscured or emphasized by the selection of appropriate scales. Testing the SEEA-EEA should include sensitivity analysis to better understand these effects.

Figure 8 A contrived example showing the two interrelated aspects of the modifiable area unit problem



Note: Box a represents the underlying data, which when grouped according to two different spatial patterns (b and c) show the same average, but different variances.

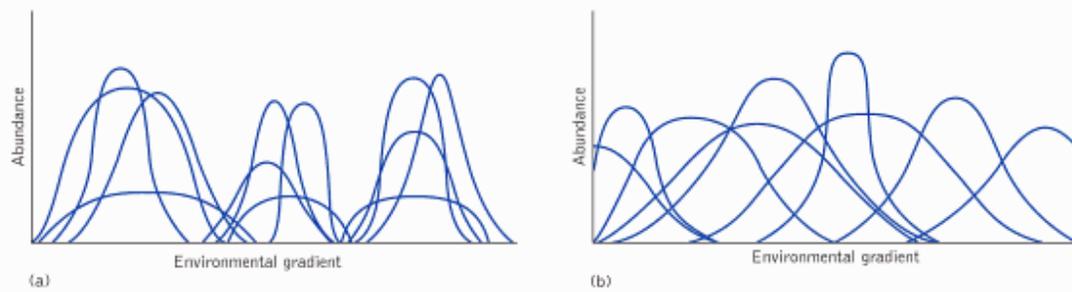
Boxes d, e and f show additional effects of using different spatial zones. Since d and e are divided into zones of the same size, their averages are retained. However, box f contains zones of different sizes, so the average value is not retained.

Source: Jelinski and Wu (1996).

it is feasible to include each individual plant in a regional scale study, it exacerbates the analysis due to excess complexity. This is akin to the approach of keeping all spatial data at the BSU level.

- *An optimal zoning approach:* This would be a zoning system that maximizes interzonal differences while minimizing intrazonal differences. This is essentially the principle behind the creation of LCEUs.
- *A sensitivity analysis approach:* This suggests additional studies on the sensitivity of variables to scale and zoning. They note that the number of variables to be analysed, the number of scales and zoning options to be tested would be large and perhaps impractical. Nevertheless, such studies as part of SEEA-EEA testing could identify appropriate scales and spatial units for relevant ecosystem attributes. Some of these studies have been

Figure 9 Two models of vegetation communities



Note: (a) represents the Clementsian model in which species' requirements coincide, leading to the identification of discrete communities. The Gleasonian model, wherein each species is distributed independently is shown in (b)
Source: Cox and Moore (2010)

conducted recently (Meynard, Lavergne et al. 2013, Kallimanis, Koutsias 2013), but there are not many in the literature.

- *Development of new methods of analysis*: This suggests more emphasis on the visualization of data. This emphasizes the earlier point about understanding the distributions of the underlying data before creating discrete classes. Many data visualization techniques are available to explore data before conducting standard statistical aggregations and analyses. Many of these are derived from Tukey's (1977) early work on exploratory data analysis. This is often used to visually identify relationships, clusters and outliers in data.
- *Emphasis of spatial analysis on the rates of change*: This is similar to sensitivity analysis except that it focusses on rates of change in variables and relationships with respect to scale. This links back to the **Coastline Problem** in that it suggests the use of fractal dimension as a scale-independent measure of spatial distribution and relationships.

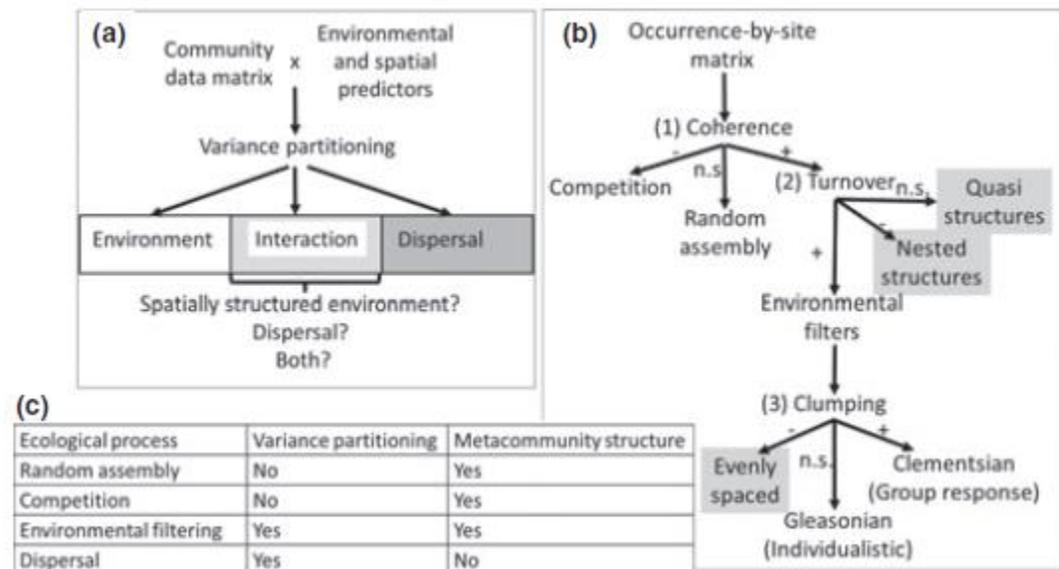
74. Jelinski and Wu (1996) conclude, "When the scale of the observational window matches the characteristic scale of the phenomenon of interest, we will see it; otherwise we miss it. These arguments form the premise of a hierarchical approach to the modifiable areal unit problem. A suggested procedure to deal with the MAUP is simply thus: first to identify the characteristic scales using methods such as spatial autocorrelation, semivariograms, fractal analysis, and spectral analysis, and then to focus the study on these scales."
75. This last statement requires some further explanation. **Spatial autocorrelation** (also known as spatial dependency or autocorrelation) refers to the fact that characteristics in close spatial proximity show higher correlations (either positive or negative) than those that are more distant (Legendre 1993). That is, the characteristics of one location can at least partially be predicted from knowing the characteristics of the surrounding area. While this may seem self-evident from an ecological perspective, it leads to the over-estimations of the correlation when standard statistical tests such as regression analysis are applied. This has led to the development of spatial analysis software that measures spatial autocorrelation and accounts for it in spatial regression models.
76. To understand the significance of Jelinski and Wu's recommendation about **semivariograms**, it is first essential to review some basic concepts in community ecology. There is an ongoing debate in ecology as to whether or not plant species cluster along environmental gradients (**Figure 9**), thus establishing discrete, identifiable communities. This model, termed Clementsian is in direct opposition to the Gleasonian model in which plant species are distributed independently with no clear communities present (Cox, Moore 2010). This is of relevance to ecosystem accounting, since

one objective is to capture changes in ecosystem condition and capacity to produce services. More research in this area would identify which drivers and conditions need to be measured as well as appropriate scales and methods for each.

Testing the SEEA-EEA should investigate the use of scale independent measures (such as variance) and appropriate methods for their treatment.

77. The traditional approach to analyse community structure is use a site-by-species incidence matrix to link observed patterns to their possible drivers. Another method is to use semivariogram functions to partition the variance of a regression analysis into factors that are explained by non-spatial environmental factors, spatial structure and unexplained variance. Meynard, Lavergne et al. (2013) compare the two approaches (**Figure 10**) at different spatial scales. They conclude that both approaches are complementary, but that variance partitioning showed stronger influences of some drivers at coarser spatial scales.
78. Linking back to the discussion of the Coastline Problem, **fractal geometry** is also used as a tool to understand the spatial patterns of ecosystems. Since ecosystems can be detailed at all spatial scales, they can be considered a fractal. That is, when zooming in from the global to microscopic level, they retain their complexity. Palmer (1988) suggests calculating the fractal dimension as the slope of the semivariogram as a single measure of spatial dependence.
79. Several authors discuss the importance of the interaction of slow and fast ecological processes in terms of resilience and thresholds (Rockström, Steffen et al. 2009, Brand 2009, Carpenter, Mooney et al. 2009). Brand (2009) further notes that resilience theory makes the controversial assumption that the slow processes control the ecosystem in terms of stability. For the purposes of ecosystem accounting, this raises questions about **temporal scaling**. Given an annual accounting period, how

Figure 10 Two approaches to assessing vegetation communities



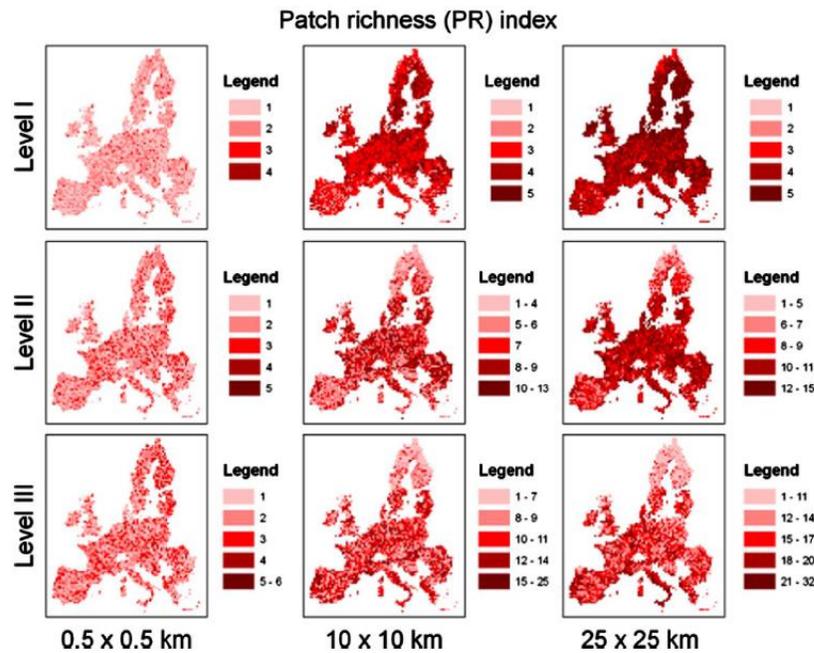
Note: (a) summarizes the framework used for variance partitioning and (b) summarizes the framework for metacommunity analysis using site-by-species analysis. The results, summarized in (c) note that each approach distinguishes different factors.

Source: Meynard, Lavergne et al. (2013).

should fast processes be distinguished and how can they scaled up and slow processes be scaled down? In general business accounting, this issue is of less relevance. Units of output, people and dollars can be added or averaged across temporal scales (daily, monthly quarterly, annual). However, in the case of ecosystems, this raises the issue of what we measure. For example, if air quality measures are taken hourly, reporting a daily average may obscure intermittent exceedances of the allowable standard (Bordt, Rastan forthcoming). In this case, a measure of the hourly exceedances would better inform policy. Understanding the variability in those exceedances would help understand the causes (e.g., traffic patterns, weather patterns) as well.

80. **Spectral analysis** is an approach that is akin to semivariograms and variance partitioning in that it partitions the variance of a time series into contributions of frequencies that are harmonics of the length of the dataset (Platt, Denman 1975). The implication is that the variance in observations can be (a) partitioned and attributed to slow and fast ecological processes and (b) represented by scale-independent measures such as variance spectra.
81. The boundary problem also permeates the collapsing of land cover types into classes (**thematic scaling**). In this case, the boundaries are the discrete classes into which a continuous distribution of vegetation densities and types are classified. Cox and Moore (2010) note that boundaries in Europe tend to be more distinct than in more recently populated areas. That is, in the Americas and Australia, *“landscapes still persist that are occupied by a continuum of vegetation rather than a clear patchwork mosaic. (p. 94)”* Without understanding the underlying distributions of the more detailed data, classifying land cover into standard categories risks underplaying the importance of this continuum. Kallimanis and Koutsias (2013) note that as spatial scale becomes coarser or as thematic resolution becomes more detailed the values of landscape diversity metrics increase (**Figure 11**). This leads to qualitatively different interpretations of the geographical locations of diversity peaks and troughs. They suggest that inferences made with low-resolution thematic classes be done with caution.
82. In collapsing land cover categories, the same principles apply as with upscaling spatial areas. That is, the remote sensing data are in reality a continuum of measures taken from different parts of the electromagnetic spectrum, such as visible light, infrared and microwave. These data are interpreted into land cover classes usually using supervised training. That is, by comparing spectral signatures with known land cover from field studies for a sample of the areas covered associations are derived that are then applied to the rest of the data. This interpretation itself introduces a source of error. The Canada Land Cover Map of 2005 contains 39 land cover categories.(Latifovic, Fernandes et al. 2014). These categories are often collapsed for reporting and analytical purposes. Statistics Canada (2013) uses 11 categories for reporting on land cover change over 2001 to 2011. This collapsing of categories is sometimes essential to address issues of inconsistency between sources of data. Pouliot, Latifovic et al. (2014) note that the level of accuracy of this collapsing of annual land cover data is on the order of 70-80%. That is, comparing the adjusted time-series with a more thematically-detailed annual reference sample showed a disagreement of 20-30%.
83. Statistics Canada (2013) communicates this uncertainty with the following statement: *“Measuring land cover categories is subject to certain limitations due to difficulties in distinguishing between different land cover types. This land cover area was derived from CCRS 250 m land cover data. Because of the coarseness of this data, areas may not be consistent with other released tabulations which used more detailed data.”*
84. In another example, Mexico’s INEGI produces land cover maps ranging from 1985 to 2011, these are based on different sources and methods. Again, comparison is done on a very broad scale of 10 classes. Mas, Velázquez et al. (2004) describe some of the approaches used to assess the accuracy of the data including comparison with known aerial photography and the national forest inventory.

Figure 11 The spatial pattern of land cover diversity in Europe



Note: Patch richness increases both with spatial scale (from 0.5 to 25km) and thematic scale (CORINE land cover classes I = 5 classes, II = 15 classes, III = 44 classes). For example, interpretations for Norway are radically different at the 25x25km scale, depending on whether the thematic scale is coarse (Norway ranks highest) or fine (Norway ranks lowest).

Source: Kallimanis and Koutsias. (2013).

85. In keeping with the principles of this section, such upscaling of thematic data is better if informed by an understanding of the underlying patterns in the source data.

Uniformity of distribution

86. While the above discussion relates mainly to **upscaling**, the importance of patterns in the data are also relevant to **downscaling**. Downscaling is the attribution of information from one scale to a smaller scale. The fourth question that every analyst considering downscaling should ask is “**Are the data uniformly distributed?**” Answering this requires some knowledge of the sources and methods used to compile the data. Some data may be sampled at lower levels and then aggregated to larger scales. For example, temperature and rainfall data are gathered from sampling stations and then upscaled to national maps, or isoclines. Other data are available only for the larger spatial unit. For example, farm production is normally reported to the NSO at the farm level. The NSO may consider this confidential and aggregate the data to the Census Agglomeration (in Canada) or other regional spatial unit. As noted in the previous section on **Spatial Units**, there may be less bias in upscaling all the data to the larger spatial unit rather than assuming uniformity by downscaling averaged data.
87. However, when it is known that the data are not uniform and analysis at the smaller spatial scale is required, for example, for conducting a local analysis, there are approaches to downscaling that consider this non-uniformity. One such approach is

If data are not uniformly distributed, downscaling those data can introduce a bias in the result. Testing the SEEA-EEA should investigate alternative data used to downscale spatial data.

probabilistic allocation based on known local conditions. This approach is often used national accounting, for example, to derive monthly GDP estimates when much of the data are derived from annual sample surveys. In the case of Canada, monthly GDP is estimated by adjusting the previous year's GDP based on monthly surveys of manufacturing, employment, payroll and hours and other survey and administrative information (Statistics Canada 2013).

88. Probabilistic allocation approaches are also used in ecological analysis to pinpoint species habitats. Most species range maps are very general and based on individual observations. Knowing the habitat preferences of species, such as vegetation type, elevation and microclimate can help narrow down the expected locations of those species. This is often systematized by the use of Habitat Suitability Index models (HSI) (Brooks 1997).
89. Such an approach could also be applied to the downscaling of complex units. For example, it is common for soil units to be represented in terms of the percentages of soil types contained within them. Similarly, large land cover units may similarly represent the mix of land cover types it represents. These could be downscaled using topographical information such as elevation and slope.

Amenability to scaling

90. As with general business accounting and national accounting, certain metrics are more amenable to scaling in ecosystem accounting than others. The fifth question each analyst should ask is “***Are my units amenable to scaling and how do I scale them?***” This issue is discussed in terms of statistical tools in and accompanying document (Bordt 2015).

The treatment of sources of error

91. No measurement operation is free of uncertainty. The sixth question every analyst should ask is “***How do I treat sources of error when scaling?***” Smith, Dick et al. (2011) note that quantifying the error associated in spatial analysis with respect to ecosystem services requires further statistical research. They suggest that the uncertainties in the underlying data are often not recognized in spatial analysis, such as in the spatial representation of air pollution concentrations without recognizing the uncertainties of the underlying models. When combining trends from data series collected from different parts of the ecosystem, they note that, “*the strength of the trend will depend on the use of a realistic spatial correlation structure in any meta-analyses, potentially requiring re-analysis of the original data series.*” This is consistent with the principles of the preceding section in that understanding the underlying uncertainties is essential to larger-scale ecosystem assessments. Smith, Dick et al. (2011) also suggest several emerging statistical approaches for analysing ecosystems and their services including methods including societal models, feedbacks and loop analysis and graphical models including Bayesian belief networks.
92. Further detail on treating sources of error is beyond the scope of this report, but this should be considered as an element of the future research agenda for the SEEA-EEA.
93. Wiens (1989) suggested that issues of scale become the primary focus of ecological research efforts: “*Instead of asking how our results vary as a function of scale, we should begin to search for consistent patterns in these scaling effects.*”

Further detail on treating sources of error should be considered as an element of the future research agenda for the SEEA-EEA. Testing the SEEA-EEA should encourage the reporting of uncertainty in all aspects of the data collection and transformation process.

Benefits transfer, value transfer and meta-analysis

94. Benefits transfer (BT) is the general name given to a range of methods that involve transferring measures from one location to another (Bateman, Mace et al. 2011, Wilson, Hoehn 2006). This can be done using studies published in the literature or unpublished case histories of the researchers.
95. Thousands of ecosystem services valuation studies over the past two decades have been codified in several online databases such as the Environmental Valuation Reference Inventory (www.evri.ca), TEEB/Ecosystem Services Partnership's Environmental Services Valuation Database ([ESVD](#)), and the WRI's Ecosystem Service Indicator Database ([ESID](#)). These codify studies into location, study type, ecosystem asset type and usually the results of the valuation study in terms of monetary value of one or more services. However, their codification of ecosystem type, conditions and services are often generalized to a few categories. A list of parameters encoded in the EVRI are included in an accompanying report (Bordt 2015)
96. **Point transfer** (termed *mean value transfer* by Bateman, Mace et al., 2011) is the simple attribution of the value of an ecosystem or its services, usually expressed in terms of average value per hectare, from one site (the study site) to an equivalent site in another location (the policy site). While this avoids the need for additional field research, it also brings with it several assumptions. The first assumption is that the biophysical and socio-economic conditions are exactly equal at both sites. Given that conditions change over space and time, this assumption is often challenged. The second assumption is that average value per hectare is a useful measure of ecosystem services production. While this measure is scalable, most analysts suggest that the marginal value (that is, the change in benefits resulting from an addition or removal of one hectare of the ecosystem asset) are more relevant to capture changing values as the stock of the resource increases or declines. Therefore marginal values are more applicable to policy contexts, where incremental changes in the stock are more relevant than the total value. However, measures of marginal value are highly context specific and thus less easily transferred.
97. A mean value obtained from a similar study site can be adjusted according to the differing conditions at the policy site. For example, a forest near a residential area may have a higher value than one that is inaccessible due to a higher recreational use. Similarly, a wetland that is absorbing nutrients from upstream farms is of a higher value to the downstream residents than one that has no nutrients to absorb. This adjustment is often done using expert judgement. Brouwer and Bateman (2005) demonstrate that when complete information on the policy site is unavailable, the results of such an adjustment can be inferior to simple mean value transfer.
98. **Meta-analysis**, referring to the use of several studies, has been applied in several ways. The first of these is a simple attribution of the averages of several studies to multiple sites, which for lack of an accepted term, we will call simple attribution. The second is function transfer.
99. Simple attribution has been applied as a variant of point transfer by Costanza et al. (1997) and others (Costanza, de Groot et al. 2014, de Groot, Brander et al. 2012, Spatial Informatics Group, Troy 2009). The approach codifies studies on service values by type of services and type of biome. Average values per hectare are averaged and applied to the total number of hectares estimated for the biome. While Costanza et al. (1997) certainly raised awareness of the importance of ecosystem services, it has been highly criticized for several reasons, including the failure to adjust for specific local conditions. That is, this approach makes the same assumptions as described above for point transfer that every hectare within any given biome provides the same level of service as the study sites. By using average values, such studies have also been criticized for perpetuating the idea of basing total asset value on average values, thereby ignoring the principle that the value of the asset changes with respect to supply and demand.

100. **Function transfer** uses the same source data, but strives to quantitatively control for the underlying factors contributing to the value reported by analysing multiple study sites. A function transfer could, for example, be based on a multiple-regression analysis of several similar sites and allocate contributions to each of the influencing factors (proximity to residential area, income of residential area, presence of upstream farms and biophysical parameters such as flow rate, vegetation density and average temperature) in the form of a transfer function. If these factors are known for the policy site, then a more defensible value of the service can be estimated by applying this transfer function. For example, Brander et al. (2006) obtained 215 value observations from 190 wetland studies, concluding that socio-economic variables such as income and population density are important in explaining wetland services (biodiversity) values. They also tested the results for out-of-sample value transfers (that is applying the results to policy sites) and found an average transfer error of 74%. They deem this comparable with errors of other value transfer exercises in the literature and ascribe the variability to the under-representation of certain types of wetlands and climatic zones in the literature and the quality of the source studies.

Testing the SEEA-EEA could contribute to the improvement existing Benefits Transfer values databases and future valuation studies by providing standard classifications of services, ecosystem types, conditions, services and valuation methods used..

101. Wilson and Hoehn (2006) concluded that, while the methods of benefit transfer have matured into viable approaches for estimating the values of ecosystem services, the quality is highly dependent on the quality and representativeness of the underlying studies.
102. Testing the SEEA-EEA could contribute to the improvement of codifying existing values databases and future valuation studies by providing standard classifications of services, ecosystem types, conditions, services and valuation methods used.
103. The statistical methods developed for BT for service values could also be applied to estimating biophysical parameters of ecosystems. That is, rather than focussing the transfer only on monetary values, similar methods could be used to impute functions and conditions that are relevant to the capacity to produce services.

Examples of scaling implementation and methods

104. Some examples were provided in the preceding section about how scaling has been implemented. This section provides a short overview and introduces some selected examples that would be informative to compiling ecosystem accounts.
105. For broad comparisons, Statistics Canada (2013) uses only three categories of “landscape type”: settled area, agricultural area and natural and naturalizing area. “Natural and naturalizing area” is the residual of the other two categories. This is done not only to simplify reporting purposes, but also to improve comparability over time.
106. Statistics Canada (2013) also reports an experimental index of water purification potential of drainage areas falling within Canada’s boreal zone. This index integrated data from various scales to the drainage area for the years 2000 and 2010 using available data and a range of scaling methods (**Table 2**). They note that the resulting index was not assessed against independent data on water quality.
107. Environment Canada, Statistics Canada and Health Canada (2007) demonstrate two further approaches to scaling. The first is that the population exposure to ground level ozone and PM_{2.5} was calculated by averaging the measures from monitoring stations weighted (scaled) by the

Table 2 Scaling methods used to calculate water purification potential

Attribute	Scale of data	Method used
Percent forested land by watershed (F)	250m land cover time series	Summed to watershed
Percent agricultural land by watershed (A)	250m land cover time series	Summed to watershed
Weighted percent riparian forest cover (R)	250m land cover time series	Calculated as the ratio between the total area of forest classes within a 250 m buffer along water bodies in a watershed and the total edge of water bodies.
Percent wetlands (W)	1:50,000 scale	Estimated as the total wetlands area summed by watershed
Percent total anthropogenic disturbance (TD)	Based on 1:50,000 LANDSAT imagery, summed to 1km grids (Pasher, Seed et al. 2013)	Estimated as the total area disturbed by polygonal (Cutblock; Mine; Reservoir; Settlement; Well site; Agriculture; Oil and Gas; Unknown) and buffered linear (road; power line; railway; seismic line; pipeline; dam; airstrip; unknown) anthropogenic features.
Weighted percent burn area (B)	Canadian National Fire Database (hectares) (Stocks, Mason et al. 2002)	Estimated by associating each watershed with the Homogeneous Fire Regime zones (HFR) in which the largest proportion of the watershed was located. Each fire regime was associated with unique scores.
Edge density (ED)	Based on 1:50,000 LANDSAT imagery, summed to 1km grids (Pasher, Seed et al. 2013)	Estimated as the total edge from linear features (road, power line, railway, seismic line, pipeline, dam, airstrip, unknown) divided by total watershed area.
Linear Density (LD)	Linear, various scales	Estimated as the total edge from linear features (power corridors, roads, railways).
Human Footprint (HF)	Linear and polygon, various scales	Estimated as the total area disturbed by settlements and linear anthropogenic features (power corridors, roads, railways) within a 1 km buffer.
Slope (S)	1:50,000 digital elevation	Derived from a digital elevation model (1:250,000) using the bilinear interpolation method of re-sampling and averaged over the watershed.
Nitrogen and Sulphur deposition exceedance level (NS)	Derived from AURAMS model. (Moran, Zheng et al. 2008)	Assigned for regions where current atmospheric deposition of N and S is greater than the critical loads ('exceeded').
Total		Each attribute was assigned a score (1-4) that was then summed. Some scores were assigned based on known ecological thresholds. If no threshold was identified, the distribution was classed into quartiles.

Note: Statistics Canada uses the term "watershed" as synonymous with drainage area or catchment area.
Source: Adapted from Statistics Canada (2013).

population living within a 40-km radius of the station. This resulted in more weight being given to highly populated areas so that the indicators are indicative of human exposure.

108. The second approach to improve the scaling of water quality data was to address the non-representativeness of Canada's water quality monitoring network. Since monitoring sites were originally chosen to identify and report on areas of concern, the distribution was not statistically representative of all surface water in Canada. Some areas, such as southern Ontario and Quebec were over-represented, while others, such as Saskatchewan, northern Ontario and northern Quebec were under-represented. By reducing the minimum criteria for inclusion for northern sites (that is,

including sites with only three samples per year rather than the four samples per year required for sites in southern Canada), more northern sites could be included. An additional methods was tested (Statistics Canada, personal communication) that would base the national average on a representative sub-sample of the monitoring sites. That is, by excluding over-sampled sites in southern Canada, the overall index would provide a provide better national statistical representation.

109. As noted in the section on **Spatial Units**, Australia's Land Account: Queensland, Experimental Estimates, 2013 (Australian Bureau of Statistics 2013) was based on cadastral units that were then overlaid with land cover data at a 250m scale. This raised concern about the possible statistical anomalies introduced by the intersection of the two. In this case, areas smaller than the resolution of the land cover data (250m) were allowed. In some spatial analysis exercises, the classification of these areas (or slivers) would have been algorithmically adjusted to that of the nearest neighbour.
110. Also described previously was Australia's approach to upscaling cadastral areas to statistical areas based on the centroid of the cadastral units. While this avoids the necessity of splitting the cadastral unit across two statistical areas, for large cadastral units, this would create accounting anomalies as well. For example, the total area of cadastral units within a statistical area would not match the known area of the statistical area. In some spatial analysis exercises, the approach is to split the cadastral unit in two and assign scaled values (area, value) to statistical areas proportionally.
111. Victoria's (Australia) approach to scaling (Eigenraam, Chua et al. 2013) was previously described in the section on **Spatial Units**.
112. The SCBD QSP data model (Weber 2014) shown in **Figure 5** suggests that downscaling be done to the BSU level and then aggregated to various spatial units for reporting. The report also includes suggestions on the treatment of uncertainty of spatial data, for example, by using Gaussian blurring to decrease the appearance of sharp transitions.

Recommendations for further testing of scaling approaches

113. The testing of the SEEA-EEA could contribute to the case history of several issues in scaling discussed in this section.
114. The implications of spatial, thematic and temporal scaling on the interpretation of the resulting indicators could be addressed by running parallel analyses. One approach could be to conduct parallel testing of different spatial scales and data sources of land cover information for the same area and period. This would help identify not only the appropriate scales, but also help identify which information was most appropriate to maintain at various spatial scales.
115. This parallel testing could also inform appropriate approaches to temporal and thematic scaling. For example, scaling daily or monthly data on ecosystem condition measures could help inform the choice of scale, but also the treatment of uncertainty. Non-linear approaches, such as semivariograms, spectral analysis and fractal analysis should be considered.
116. Another area that deserves further attention is the treatment and reporting of uncertainty in the interpretation of spatial information. This could also be addressed, in part by parallel testing of different methods of interpretation. Various approaches to interpretation are discussed in the literature, but there seems to be no generally accepted approach to reporting on this error.
117. Testing of the SEEA-EEA should encourage the reporting of uncertainty in all aspects of the data collection and transformation process. Guidelines on calculating, documenting and using uncertainty measures would assist compilers and users alike in providing a better understanding of the reliability of the underlying data and aggregates. One test that could be done would be to

analyse the uncertainty in the underlying data contributing to various indices and how this impacts the outcome.

118. Scale-independent measures (such as variance) and their treatment should be further investigated.
119. Best practices could also be developed around downscaling. For example, what data could be used to allocate specific measures to smaller spatial scales?
120. Benefits transfer approaches should be tested, but not necessarily with the objective of transferring monetary values. Methodologies developed for function transfer of ecosystem services benefits could also be applied to imputing biophysical measures and the levels of uncertainty in this imputation.
121. One generally-applicable tool that could be developed is a framework for recording valuation studies. A standardized approach to codifying location, ecosystem type, condition measures, socio-economic conditions, ecosystem service and valuation method used would benefit researchers conducting these studies as well as users of the data.
122. Another tool that could be developed would be to codify the individual CICES services in terms of the scale of the service and the scale of the beneficiary. This would complement codifying the CICES services in terms of ecosystem types and linking to ecosystem functions suggested in **an accompanying report (Bordt 2015)**.

Amenability to official statistics

123. One role of NSOs as compilers of ecosystem accounts is to ensure and improve the statistical integrity of the resulting accounts as data are upscaled, downscaled and transferred. Most NSOs employ statistical methodologists to assure quality and to advise on the appropriate treatment of data. However, these methodologists may not be familiar with the nature of the data and methods used to compile spatially-referenced data on ecosystems.

One role of NSOs as compilers of ecosystem accounts is to ensure and improve the statistical integrity of the resulting accounts as data are upscaled, downscaled and transferred. The objective should not only be to treat the uncertainty in the environmental data, but to encourage those collecting the environmental data in other agencies to improve the statistical rigour of their work.

124. The objective should not only be to treat the uncertainty in the environmental data, but to encourage those collecting the environmental data in other agencies to improve the statistical rigour of their work. Joint learning would be enhanced by including methodologists in the project team for compiling pilot ecosystem accounts.

3.3 Aggregation

125. Aggregation refers to the process of combining several measures into a simpler set of measures. Understanding how and when to aggregate information is important not only to summarizing the information for communication purposes, but also for analytical integrity. In terms of communication, information is often aggregated into indicators, indices and graphics. For analytical purposes, disparate measures (e.g., ecosystem service flows for flood control and wildlife habitat) may be aggregated using a “common currency”, such as the market value of the services or by creating an index based on a reference state. Furthermore, they may be aggregated using common units of measure such as tonnes or CO₂ equivalents.

Representation in SEEA-EEA

126. The SEEA-EEA addresses aggregation at several levels:

- The aggregation of spatial units (para 2.51): This topic is addressed in this report in the sections on **Spatial Units** and **Scaling**.
- Aggregation of ecosystem condition (para 2.95, para 4.86): This suggests the creation of an index based on a reference state. This is flagged in the research agenda for further research. Measures of condition, choice of reference state and related issues are discussed in Deliverable 2.a.1.
- Aggregation across different ecosystem services (para 2.88, 3.65): This notes the importance of understanding the inter-relationships between services (competing, independent or tandem), their relative importance (weighting) and opportunities for converting to common units, such as composite indices and equivalent units.
- Aggregating future ecosystem services to provide an estimated stock of future ecosystem service flows (para 2.31, 4.85): This suggests creating a monetary or non-monetary aggregate across all expected future ecosystem service flows. This is also flagged in the research agenda for further research.
- Providing aggregate information for measuring trends and comparing ecosystem assets for policy and analytical purposes (para 4.4): This suggests that the SEEA-EEA should identify the most relevant aspects of ecosystem assets.
- Aggregation for ecosystem accounting in monetary terms (para 5.112): This covers valuation issues and is not discussed in detail in this section, as it will be detailed in a separate sub-document to the Guidance Document.

Selected issues in aggregation

127. This section will focus on advancing our understanding on selected issues within the above scope with the intent of (a) reviewing existing approaches and (b) identifying objectives for the testing of the SEEA-EEA:

- **Aggregating biophysical measures of ecosystem condition and capacity:** Measures of ecosystem condition (themselves aggregates) are varied and vary across ecosystem types. For ecosystem accounting purposes, ideally these could be combined into a single composite ecosystem condition index that would be responsive to changes in ecosystem quality. This section discusses the selection of such a reference state.
- **Aggregating biophysical measures of services:** Flows of services are initially measured in biophysical terms and, like condition measures, are ideally combined into a single measure, or small set of measures. These disparate measures could be aggregated using a common currency or an index, much like ecosystem condition. While examples of both approaches exist, there is little agreement on how to apply them to ecosystem accounting. There is also the argument that some services are so critical to human existence that they should not be monetized or combined with other measures (e.g., oxygen and biomass production), but there is little agreement among scientists on what these fundamental values actually are (Brand 2009, Jax, Barton et al. 2013). This work will be reviewed and examples of implementation provided.
- **Aggregation by creating a composite index:** This is suggested in the SEEA-EEA as an approach for both ecosystem condition and services. Some examples and considerations are discussed below.
- **Producing final aggregates that are applicable to various decision contexts:** The SEEA-EEA does not recommend a single final aggregate (such as GDP from the economic accounting). However, there is a need to summarize the complex information

contained in an ecosystem account for decision makers. Attention must therefore also be paid to the purpose of the aggregate. Resource management, economic decision making and conservation, for example, will benefit from different aggregates to appropriately address the decisions with which they are faced. Work on multi-criteria decision making will be reviewed.

Examples of implementation

The selection of a reference state

128. One approach to aggregating biophysical measures of condition is to compare the current state with a reference state. That is, the current value of each condition measure is indexed with respect to a benchmark for that measure. The resulting indices, for example scaled to 0 to 100, are then aggregated. Such reference states, or benchmarks, can represent a specific time, a known “ideal” condition or a theoretical condition. It is important to note that reference states are not necessarily the same as the target conditions. For example, agricultural landscapes would not be judged in terms of their deviation from a pristine state, but in terms of their deviation from state of sustainable production.
129. In determining reference states for each element in Norway’s Nature Index (NNI), Certain and Skarpass (2010) provides a useful starting point. The Nature Index is intended to reflect the state of biodiversity and not necessarily a broad range of ecological condition measures. Certain and Skarpass (2010) suggest that the contributing experts select from the most practical of:
 - **Carrying capacity:** “*A theoretical value for a population number or density for example, according to the natural limit of a population set by resources in a particular environment.*” This refers to the capacity of the ecosystem to support a specific number or density of a particular species. This is appropriate for species-specific information, but not easily transferrable to other types of information on ecosystem condition.
 - **Precautionary level:** “*Recommendations provided by scientific and independent group of reflexion. Refers to a value below which the indicator, and therefore the major habitat to which it is related, is endangered.*” That is, a recommended level based on scientific evidence. This could be transferred to other indicators, such as levels of pollution and degrees of disturbance. Farmer and Randall (1998) suggest the concept of Safe Minimum Standard as a useful balance between risk aversion and economic efficiency.
 - **Pristine or near pristine state:** “*An estimated value that refers to pristine, untouched or low impacted natural system.*” This, like Carrying Capacity, is also a theoretical value. Mexico’s INEGI (Mas, Velázquez et al. 2004), South Africa (Driver 2012) and Victoria, Australia (Eigenraam, Chua et al. 2013) have based land cover or vegetation change on estimated or modelled pre-development conditions. Pristine reference states for other condition measures could also be estimated, for example for freshwater, air and habitat quality.
 - **Knowledge of past situation:** “*An estimated value derived from a known past situation, when the indicator was in good condition, and a situation that is always ecological relevant today.*” This is also intended to reflect species numbers and density, but could apply to other condition measures such as freshwater and soil quality.
 - **Traditionally managed habitat:** “*A value observed under traditionally managed habitat, such as extensive, biological agriculture.*” This is an example of the distinction between reference states and target condition. For example, even under extensive use, this reference state could be set to reflect an optimal flow of services.
 - **Maximum sustainable value:** “*A value below which no detrimental effects are observed for the major habitat to which the indicator is related.*” That is, higher values of species

numbers or densities would imply a negative impact on the habitat. This may only be applicable for more abundant and invasive species. The reference condition could be applied more generally to condition indicators, for example, in terms of the capacity of ecosystems to assimilate pollutants (buffering capacity).

The practical choice of reference states for ecosystem accounting requires further research through testing and literature search. The examples investigated suggest that a pristine state (or pre-development state) may be a good benchmark.

- **Best theoretical value of indices:** “If the indicator refers to an already developed index, such as a biodiversity index, it’s best (the value corresponding to the “best” state in term of biodiversity) expected value depending on the location and the major habitat.” While this is intended for biodiversity indices, it could also be used for other holistic indices of ecosystem quality (see discussion of ecosystem health and eco-exergy in **Deliverable 2.a.1**).
 - **Amplitude of fluctuations observed in the past:** “For fluctuating populations (typically rodents or small pelagic fishes): the amplitude of fluctuations over a given temporal windows that is observed in natural or low impacted conditions (specific case for pristine or past knowledge).” This is consistent with the discussion in **an accompanying report (Bordt 2015)** in terms of the variance in certain indicators being a predictor of ecosystem regime shifts.
130. Reference conditions are sometimes represented in terms of the earliest available comparable data. This is the main approach used by Statistics Canada (2013). This is also consistent with the suggestion in the SEEA-EEA that one option for the reference condition simply be the beginning of the accounting period. It is also more defensible in terms of official statistics in that it requires fewer assumptions (about the past, estimating theoretical values, etc.). However, basing targets on arbitrary past conditions risks undermining the objective of managing ecosystems more sustainably. For example, short-term, recent improvements in ecosystem condition may be seen as progress when in fact the improvements may be negligible in terms of setting realistic targets for achieving more sustainable conditions.
131. The SCBD QSP (Weber 2014) suggests several approaches to establishing reference conditions. One is the NNI approach above, another is the approach used by Cosier and McDonald’s (2010), which is the condition of Australia at the time of its discovery by Europeans.
132. As discussed in an accompanying report (**Bordt 2015**), reference conditions have been established for air and water quality indices in terms of standards that should not be exceeded relative to human health or specific use (Environment Canada, Statistics Canada & Health Canada 2007). Such standards have been applied by Statistics Canada (2013) and Malouin, Doyle et al. (2013) in the compilation of the water purification potential indicator described in the **Scale and Scaling** section. In this case, this was done by calculating the number of times the nitrogen and sulphur deposition rates exceeded critical loads.
133. The practical choice of reference conditions for ecosystem accounting requires further research through testing and literature search. The above examples suggest that a pristine state (or pre-development state) may be a good benchmark. However, this needs to be distinguished from an achievable target state.
134. Statistical issues in aggregation are discussed further below in the discussion on creating indices.

Aggregating biophysical measures of ecosystem services

135. Other than valuation in monetary terms, there has been less experience in aggregating biophysical measures of ecosystem services into a single index. Monetary valuation alone imposes limitations in that only certain services (e.g., the contribution of ecosystems to economic production such as crops or timber), while others may be represented by incomplete proxies (e.g., using flood insurance costs as a proxy for the contribution of ecosystems to flood control). This section reviews some approaches to analysing “bundles of services”, that is, aggregating similar services within categories, thereby avoiding aggregation into a single composite index. This section also reviews the current debate regarding “critical services” in that certain services may be fundamental to well-being and should therefore not be aggregated or monetized.
- Ecosystem services can be analyzed in “bundles” that are closely associated with each other. This would avoid the need for aggregating ecosystem services into one composite index.*
136. Aggregating biophysical measures of ecosystem services could be accomplished by comparing the flow of each service with a reference condition as well. Certain caveats have been noted in the SEEA-EEA in terms of the relationships between services. That is, some will compete with one-another and others occur in parallel. An increase in one service may cause the decrease in another (for example, increasing agricultural intensification may decrease habitat potential). Other services may be linked (for example, improving wetland quality could improve flood control and water purification potential).
137. Understanding the linkages between services would be supported by the recommendation in an accompanying report (**Bordt 2015**) to conceptually map the production of services with ecosystem condition and important ecosystem processes. Rather than aggregating services, bundles of services

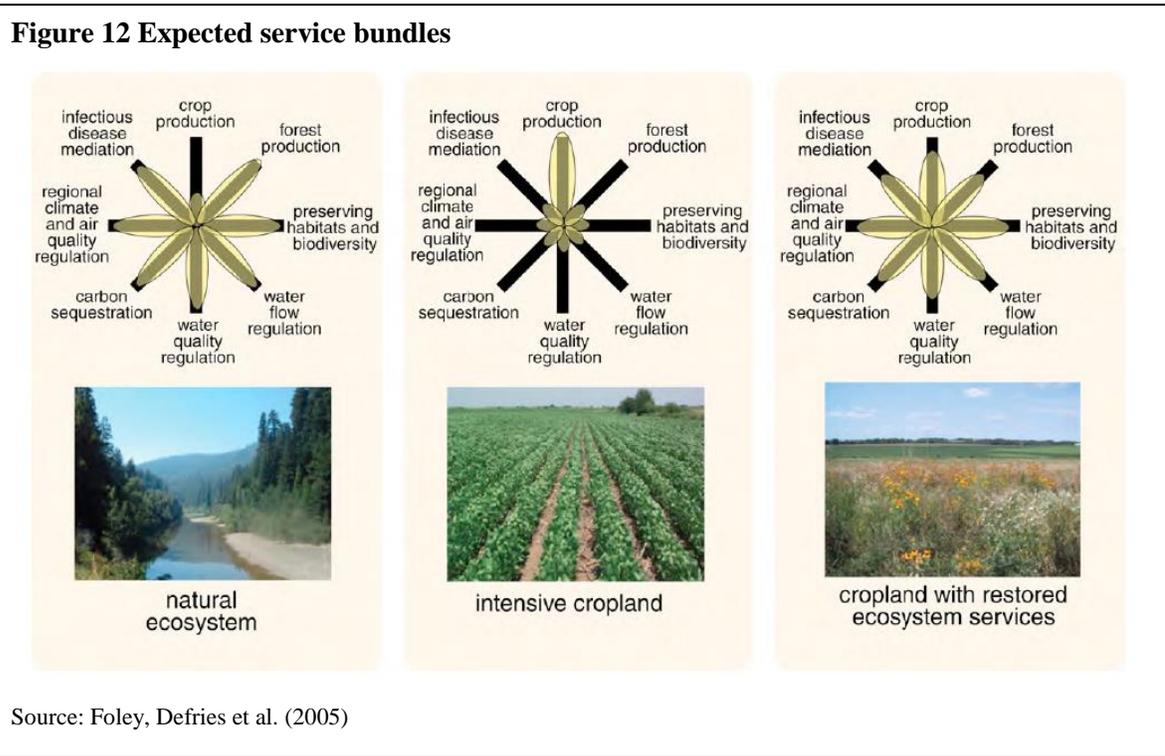
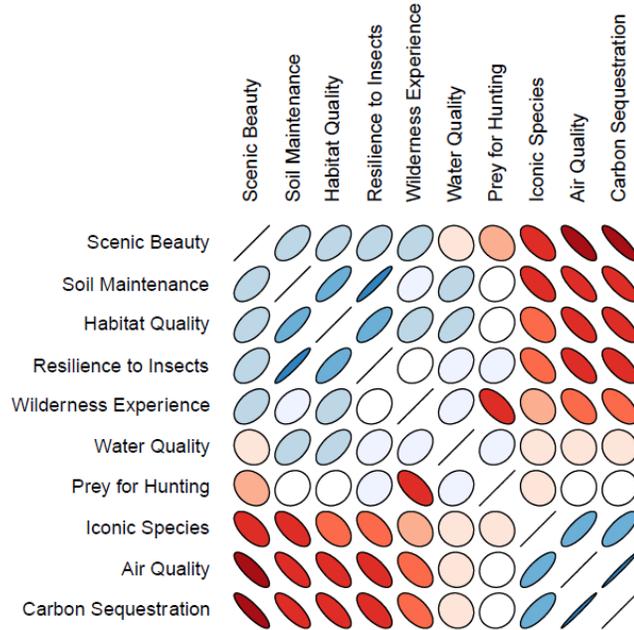


Figure 13 Average service potential across Ecoprovinces



Note: Darker colours and narrower ellipses indicate stronger correlations. Blue colouring indicates positive correlations, red colouring indicates negative correlations.
 Source: Malouin, Doyle et al. (2013)

(Figure 12) are often shown individually across different scenarios (Foley, Defries et al. 2005, Raudsepp-Hearne, Peterson et al. 2010). This constitutes a multi-criteria context which is discussed further below in terms of **final aggregates**.

138. Malouin, Doyle et al. (2013) demonstrate a correlation approach to establishing bundles of services (Figure 13). They note that two distinct bundles emerged: (1) Soil Maintenance, Habitat Quality and Resilience to Epidemic Insect Outbreaks and (2) Habitat for Charismatic or Iconic Species, Air Quality, and Carbon Sequestration. They note that these two bundles appeared to be negatively correlated with each other. That is, there appear to be tradeoffs between the two bundles. There also appeared to be negative correlations between individual services, for example (a) Scenic Beauty with Carbon Sequestration and (b) Prey for hunting with Wildlife experience.
139. Some reference conditions for individual provisioning services could be derived from existing work on resource management, such as maximum sustainable yield (MSY) in fisheries (Maunder 2002) and optimal harvest in forestry (Gassmann 1989). Regulating and maintenance and cultural services could be benchmarked to optimal or past reference conditions.
140. The weighting of services, that is, assigning a measure of importance to each is also suggested in the SEEA-EEA. However, as noted by several authors (Ayres 2007, Brand 2009), certain services may be of such critical importance that they should not be monetized or included in any composite index of other services that may be more substitutable. Ayres (2007) notes that there are no substitutes for natural processes in the creation of oxygen, biomass and certain

Some reference states for individual provisioning services could be derived from existing work on resource management, such as maximum sustainable yield (MSY) in fisheries and optimal harvest in forestry. Regulating and maintenance and cultural services could be benchmarked to optimal or past reference states.

nutrients essential to humans. Brand (2009) suggests that since future conditions are unknown, ecological resilience itself is a critical natural capital. While this relates more to the condition of ecosystems, the role of regulating and maintenance services in maintaining ecosystem resilience is not well understood (See Deliverable 2.a.1). In addition, Jax, Barton et al. (2013) view certain provisioning and cultural services ecosystem services (such as fish harvest, existence of certain ecosystems in terms of their capacity to transform people's relationship with nature) as fundamental to the identity and way of life to beneficiaries. This suggests more attention to the relative importance of services, not only to the economy, but also to long-term human and ecological well-being.

Rather than weighting ecosystem services a priori, guidance could be provided on establishing appropriate weighting schemes across subsets (bundles) of services.

141. One approach taken to weighting of environmental impacts is that taken by some approaches to Life Cycle Impact Assessment (Goedkoop, Spriensma 2001) wherein damage scores on human health, climate change, resource stocks and ecosystem quality are weighted using a panel approach. The results are sometimes presented in terms of different aggregates (scenarios) depending on which social values take precedence (such as priority on human health). Such an approach could also be adapted for the purposes of aggregating ecosystem services. That is, rather than weighting ecosystem services *a priori*, guidance could be provided on establishing appropriate weighting schemes across subsets of services (food, water, materials, energy, mediation of waste, mediation of flows, maintenance of conditions, cultural).
142. Statistical issues with respect to aggregating services are included below in the discussion on creating indices.
143. The SCBD QSP (Weber 2014) focuses on two types of aggregates of ecosystem services. The first is Accessible Ecosystem Infrastructure Potential, which is expressed in terms of stocks of landscape ecosystem potential and stocks of river ecosystem potential. Since these are indices, they can be added. The second type of service aggregate is Access to Ecosystem Infrastructure Potential, which records the local, basin, regional and global access to Total Ecosystem Infrastructure Potential. Different approaches are suggested for calculating access at each scale, but the result is essentially a population-weighted measure.
144. Statistics Canada (2013) tested the aggregation of biophysical ecosystem services in terms of the potential to provide services. One example for water purification potential was shown previously in **Table 2**. Other services were treated using the same approach (see **Figure 13**). Details were reported by Malouin, Doyle et al. (2013). Rather than aggregating across different service potentials over spatial units, the authors suggest reporting on the service potentials individually in the form of a flower diagram (**Figure 14**).

Creating indices

145. As suggested in the SEEA-EEA, measures of condition and services can be aggregated into a single index by indexing each element to a reference state. In preparation for creating an index of **ecosystem condition**, several considerations should be taken into account:
 - Is there an accepted approach to assign a relative importance (weight) of each component indicator? For example, in indexing the condition parameters suggested in SEEA-EEA Table 4.3, all five characteristics could be assigned an equal important (weight) or each could be assigned an individual weight based on an agreed scheme. Ideally, this weight would be based on knowledge about the links between the condition and the capacity of the ecosystem to produce a desired bundle of services.

Figure 14 Average potential to provide 10 services – Great Bear Lowlands Ecoprovince



Source: Malouin, Doyle et al. (2013)

- Is the reference state meaningful and measurable? The Norway Nature Index discussed above provides a range of optional reference states, depending on availability and the type of ecosystem. Furthermore, the reference state needs to be distinguished from a target or ideal condition. For example, a cultivated cropland, despite being highly modified, produces a range of services. The ideal condition would not be to return the cropland to pre-development conditions, but to manage the cropland in a way that reduces further environmental degradation. As discussed above, selecting an arbitrary or recent reference state could bias the interpretation.
- Is the indicator a quality measure or a classification measure? For example, some condition measures are physical parameters (such as elevation or streamflow) that do not necessarily reflect changes with respect to ecosystem quality.
- Does an increase in the measure imply an increase or decrease in ecosystem quality? It is important to normalize all measures to the same direction and scale. For example, an increase in a level of contaminant in freshwater would imply a decrease in quality. However, generally a decrease in diversity or species richness would be taken as a decrease in quality. For many indicators, any divergence (increase or decrease) from the optimal condition will imply a reduction in quality.
- Does the measure relate to the capacity for the ecosystem to produce services? Some condition measures are more strongly associated with capacity to produce services than others. For example, to understand the capacity for a wetland to remove metals, the type

of vegetation and flow rates are important. Other condition measures, for example, air quality may be less relevant.

- Are the measures independent? If there are many interdependencies between measures. For example, increasing phosphorous in freshwater may result in increasing biomass. If phosphorous levels and biomass are both used in the index, then they will be over-represented in the index since both will change in the same direction. One approach to reducing the internal correlations would be to conduct a principal component analysis on the candidate measures. This is similar to the problem of spatial autocorrelation discussed in the **Scale and Scaling** section.
- Are the measures amenable to scaling and aggregation? For example, categorical variables, such as soil class, may be represented as numerical variables, but a change in the value does not imply an equivalent change in condition. Binary (yes/no) and ordinal classes (1=poor, 2=fair, 3=good, 4=excellent) of condition should also be treated with caution since small changes in the underlying condition could cause a change in class that would have a disproportionate impact on the index. Measures may have the same units, but different impacts. Greenhouse gases are first converted to CO₂ equivalents before being added together. Is there an accepted approach to establishing sub-indices? Sub-indices are sometimes created to aggregate specific groups of indicators, such as all indicators relating to water quality. Indonesia's Experimental Environmental Quality Index (Airlangga 2013) contains weighted sub-indices for air quality, water quality, land habitat quality and population density⁸. The Canadian Water Quality Index (Environment Canada, Statistics Canada & Health Canada 2007), sets thresholds for each substance in terms of a particular use of water: drinking, recreation, irrigation, livestock watering, wildlife and aquatic life. The composite indicator then counts the number of parameters that have exceeded these thresholds. Sites are classified into categories of poor, marginal, fair, good and excellent for each use. The counts of sites within each category can then be aggregated to drainage area and compared over time.

146. A simple composite index (e.g., adding the component indicators) runs the risk of assuming that the component parts are substitutable. For example, an increase in the population of one species would be shown to compensate for a decrease in the population of another. In a broad composite index, a decrease in water quality may be compensated for by an improvement in air quality. Without access to the underlying component indicators, the interpretation may be that there is no change. This issue has been addressed in human health indices by indexing each component indicator in terms of its impact on human health (Stieb, Burnett et al. 2008).
147. The same considerations can be applied to creating an index of ecosystem services. However, issues of weighting, scaling and independence are less well understood. Rather than focussing on ecosystem quality, an index of ecosystem services would ideally reflect the changing contributions of ecosystems not only to the economy, but to other aspects of human well-being as well.

Producing final aggregates

148. Producing final aggregates beyond monetary ones requires consideration of human well-being, time horizons, social preferences and decision contexts. Wealth accounting (Arrow, Dasgupta et al. 2010) and ecosystem health perspectives (Jørgensen, Xu et al. 2010, Rapport, Costanza et al. 1998) are largely based on the premise that maintaining this wealth, or the health of ecosystems, would ensure a constant flow of services. However, since ecosystems are being converted and degraded

⁸ This is based on the [Virginia Environmental Quality Index](#), which also contains components for toxic releases, breeding birds, forests and wetlands. Such indices should be further investigated for their application to ecosystem accounting.

faster than their capacity to regenerate (MA 2005), simply tracking ecosystem condition is insufficient. One objective of ecosystem accounting is to identify the most relevant aspects of ecosystem assets, their condition, their capacity to produce services and the flow of services from them to provide a coherent set of evidence to inform decisions about trade-offs between development and conservation.

A “dashboard” of key indicators that reflect the state and changes in state of important ecosystem conditions and services would better reflect the complexity of ecosystems than a single composite indicator.

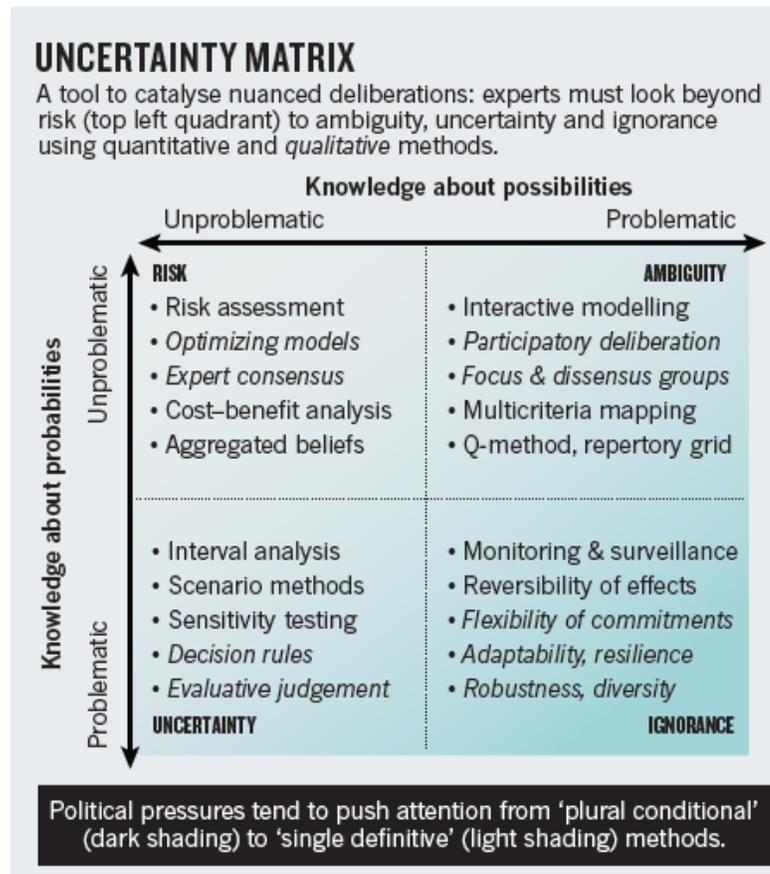
149. Ideally, this evidence would go well beyond monetary considerations and also inform decisions about trade-offs for long-term human well-being. The Millennium Ecosystem Assessment (2005) provides a starting point for components of well-being (security, the basic material for a good life, health, and social and cultural relations). The linkages between ecosystem services and these components are not well quantified, although Kandziora et al. (2012) provide a useful literature review and conceptual framework (**Figure 15**). While the objective of their paper was not to establish aggregates, it does suggest a means for broadening the scope of such aggregates.
150. As noted earlier, existing implementations of ecosystem accounting focus on aggregates of ecosystem condition or potential service provision for individual services rather than aggregates of

Table 15 Direct interrelations between selected ecosystem services and some human well-being criteria

Human well-being indicators (X) ↕ Ecosystem services (Y)		Economic attributes of well-being					Social attributes of well-being					Personal well-being	
		Income	Employment	Housing	Infrastructure	Security	Nutrition	Demography	Health	Education	Leisure		Social relations
Regulating services	9 Global climate regulation					↗	↗	↗	↗				↗
	10 Local climate regulation			↗		↗			↗				↗
	11 Air quality regulation					↗			↗				↗
	12 Water flow regulation			↗	↗	↗			↗				↗
	13 Water purification					↗	↗		↗				↗
	14 Nutrient regulation					↗	↗		↗				↗
	15 Erosion regulation					↗	↗		↗				↗
Provisioning services	16 Natural hazard protection	↗	↗	↗	↗	↗	↗	↗	↗	↗			↗
	20 Crops	↗	↗			↗	↗	↗	↗				↗
	21 Biomass for energy	↗	↗	↗	↗	↗							↗
	22 Fodder	↗	↗			↗	↗						↗
	23 Livestock	↗	↗			↗	↗	↗	↗				↗
	24 Fibre	↗	↗					↗	↗				↗
	25 Timber	↗	↗	↗	↗	↗					↗		↗
	26 Wood fuel	↗	↗		↗	↗			↗		↗		↗
	27 Fish and seafood	↗	↗			↗	↗		↗		↗		↗
	28 Aquaculture	↗	↗				↗		↗				↗
	29 Wild foods	↗					↗		↗		↗		↗
Cultural services	30 Biochemicals	↗	↗				↗		↗		↗		↗
	31 Freshwater			↗	↗	↗	↗	↗	↗	↗	↗		↗
	34 Recreation & tourism	↗	↗	↗	↗			↗	↗	↗	↗	↗	↗
	35 Landscape aesthetics	↗	↗						↗		↗	↗	↗
	36 Knowledge systems	↗	↗						↗	↗	↗	↗	↗
	37 Religious experience										↗	↗	↗
	38 Cultural heritage	↗	↗					↗	↗	↗	↗	↗	
	39 Natural heritage	↗	↗		↗	↗		↗	↗	↗	↗	↗	

Source: Kandziora et al. (2013).

Table 16 Stirling’s (2010) Uncertainty Matrix



Source: Stirling (2010).

- all services. The SCBD QSP (Weber 2014) suggests a balance sheet showing changes in ecosystem capital in terms of ecological debts (degradation) and credits (restoration) in terms of ECU (ecosystem capability units).
151. Rather than focussing on one aggregate for all purposes, it may be more productive to provide a “dashboard” of different aggregates for different decision contexts that take into account social preferences. Smith et al. (2011) argue that there is a need for “*multi-criteria analysis allowing for use of both quantitative and qualitative data and searching for a balance between several objectives set on different criteria.*” Stirling (2010) suggests that over-simplifying complex and uncertain information for the purposes of decision making is misguided. His approach is to apply appropriate methods in various contexts of uncertainty to establish science advice that takes into account multiple viewpoints and alternative future conditions (**Figure 16**).
 152. For ecosystem accounting aggregates to be useful, they need be able to support alternative views of the state of the real world and technological progress. The Millennium Ecosystem Assessment developed four scenarios (Carpenter, Bennett et al. 2006) to deal with not only the uncertainty in the evidence, but also

Stirling suggests that oversimplifying complex and uncertain information for decision making is misguided. Appropriate methods should be applied depending on our completeness of knowledge and the uncertainty of the phenomena we are measuring.

uncertainty about the future:

- Global Orchestration: global economic liberalization with strong policies to reduce poverty and inequality, and substantial investment in public goods such as education.
- Order from Strength: economies become more regionalized, and nations emphasize their individual security.
- Adapting Mosaic: more regionalized economies, but there is emphasis on multi-scale, cross-sectoral effort to sustain ecosystem services.
- TechnoGarden: the economy is globalized, with substantial investments in sound environmental technology, engineered ecosystems, and market-based solutions to environmental problems.

153. The scenario approach to assessing the impacts of differing levels of services was also used by the UK National Ecosystem Assessment (UK DEFRA 2011).

154. This is not to suggest that ecosystem accounting should incorporate this broad scope of human well-being, time horizons, social preferences and decision contexts. However, being aware of (a) the uncertainties of the underlying information and (b) the opportunities for informing a wide range of uses should support the determination of “*relevant aspects of ecosystem assets.*”

Recommendations for testing

155. Testing the SEEA-EEA would benefit from the development of certain tools that would support the determination of reference conditions and priorities among (and linkages between) ecosystem services:

- Further testing the SEEA-EEA could investigate the use of a “dashboard” of several key aggregates (e.g., ecosystem condition sub-indices, services bundles) that would communicate the complexity of changes in ecosystems to a variety of decision contexts.
- A compilation and codification of actual reference states for ecosystem condition and services: Several could be derived from water quality and air quality standards. However, for other ecosystem conditions and services, a more extensive literature survey would be required. Cosier and McDonald (2010) provide a set of references used for their *econd* index.
- A survey of ecosystem service priorities: To support the aggregation of ecosystem services, it would be useful to have a more detailed survey of experts on the priority of ecosystem services. Such a survey was conducted with the intent to establish feasible services for quantification in ecosystem accounting (Hein 2012), but this could be expanded to include a wider range of experts and a more conceptual perspective on the importance of services. This could support the development of weights for aggregation. By including decision makers as well, it could also establish priorities for reporting on different decision contexts.
- Both suggestions above, could support the testing of the SEEA-EEA by linking directly to the CICES. The nature of the recommended benchmark (optimal yield, past condition, sustainable level) and level of criticality could be recommended for each service.
- A review of how various ecosystem services models treat aggregation and the final aggregates they present would also provide valuable insights.

Amenability to official statistics

156. As with the section on **Scale and Scaling**, the NSO will likely have the methodological expertise to support the aggregation of ecosystem condition and services data. As well, the compilation of

indices would benefit from their support in conducting and interpreting statistical procedures such as principal component analysis, normalization and weighting.

4. Further work

157. Recommendations for further work in terms of testing the SEEA-EEA are discussed in each section.
158. The report would benefit from review and additional input from experts in the areas discussed.

5. Links to further material

See References.

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7. Annex 1: Compilation guideline for spatial units

This Annex provides guidance for compiling spatial units. The objective of creating coherent spatial units is to consolidate ecosystem information (ecosystem type, condition, services) at appropriate levels of spatial scale. The delineation of spatial units requires a Geographic Information System (GIS) and appropriate skills to perform this compilation.

7.1 Choice of Basic Spatial Unit (BSUs)

The choice of BSU will depend on data availability and the level at which data in the account need to be analyzed.

Minimum data required for the delineation of a BSU is land cover at the national level for at least two periods (years) derived from satellite imagery.

Ideally, the land cover classification would be based on the SEEA-EEA land cover classification (**Figure A1**). If more thematic detail is available, then these categories should be classified with respect to the 16 high-level categories of the SEEA-EEA. Note that this classification includes wetlands, inland and coastal water bodies and sea.

The size of the BSU should be the maximum resolution available from satellite imagery. For example, if data are available at 30m resolution, this should be used.

In some countries, national, detailed land cover information may already be available and in GIS format. If not, global land cover data are available from FAO at 1km resolution. If there is no consistent land cover information available for your country at a more detailed scale, this could be used as a starting point. However, the choice of a lower resolution will reduce the utility of the ecosystem account for the analysis of local conditions.

If the entire ecosystem account is to be maintained within a GIS, and all contributors have the required skills to manipulate these data, then creating LCEUs may not be necessary for certain types of analysis, such as land cover and land cover change summary tables.

7.2 Delineating Land Cover Ecosystem Functional Units (LCEUs)

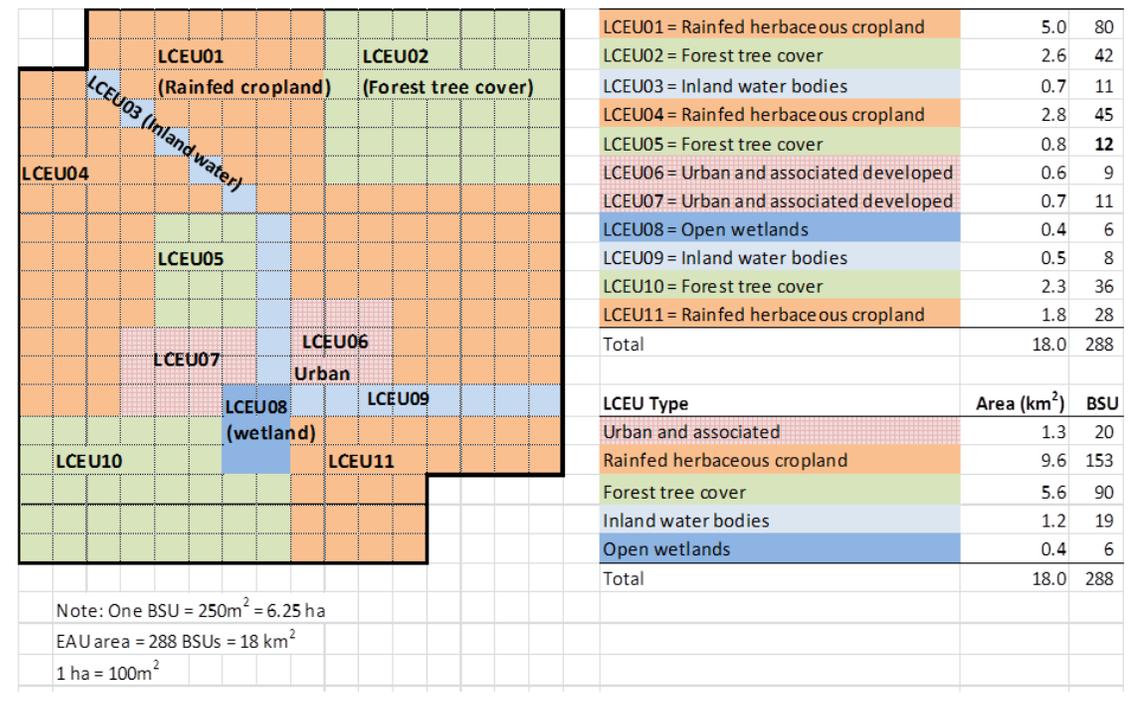
The objective of creating LCEUs is to create “optimal zones”, or areas that are homogenous for the purposes of the analysis. As with BSUs, the minimum requirements are land cover information for two periods. An LCEU may be minimally defined as a contiguous area of the same land cover.

Figure A1 shows an area with 288 BSUs that have been aggregated by land cover type into 11 distinct LCEUs. For example, LCEU01 and LCEU04 are both classified as “rainfed herbaceous cropland”, but since they are separated by a river (“inland water bodies”), they are distinct LCEUs.

Figure A1: SEEA-EEA Provisional land cover classification

- Urban and associated developed areas
- Medium to large fields rainfed herbaceous cropland
- Medium to large fields irrigated herbaceous cropland
- Permanent crops, agriculture plantations
- Agriculture associations and mosaics
- Pastures and natural grassland
- Forest tree cover
- Shrubland, bushland, heathland
- Sparsely vegetated areas
- Natural vegetation associations and mosaics
- Barren land
- Permanent snow and glaciers
- Open wetlands
- Inland water bodies
- Coastal water bodies
- Sea

Figure A1 SEEA-EEA recommended spatial units



Additional information may be used to improve the homogeneity of LCEUs. For example, if one part of a “forest tree cover” LCEU is used for forestry and another part is a protected area, then it could be divided into two LCEUs, if information on land use is available.

Other information that could be overlaid onto land cover to increase the homogeneity of LCEUs includes: land use, barriers (roads, power transmission lines, etc.) ownership (cadastral data), hydrology (data on rivers, streams and wetlands may be available from other sources and could provide more detail than satellite imagery), topography (elevation, slope) and soil type.

7.3 Ecosystem Accounting Units (EAUs)

For reporting, LCEUs are further aggregated into EAUs (Ecosystem Accounting Units). EAUs can be based on drainage area (catchment area), ecological units (e.g., ecozones) or administrative areas (e.g., conservation areas, county, state).

Since EAUs are aggregates of LCEUs, it is important that LCEUs do not cross EAU boundaries. For example, if the EAU is to be defined as a drainage area and an LCEU crosses two drainage areas, then it should be further divided into two parts along the drainage area boundary (height of land or watershed).