Journal of Environmental Management 126 (2013) 13–19

Contents lists available at SciVerse ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Scale dependency of biocapacity and the fallacy of unsustainable development

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ARTICLE INFO

Article history:
Received 3 August 2012
Received in revised form
1 February 2013
Accepted 6 April 2013
Available online 3 May 2013

Keywords:
Biocapacity
Scale
Resolution
Jinghe River Watershed
Shiyang River Basin
Modifiable areal unit problem

ABSTRACT

Area-based information obtained from remote sensing and aerial photography is often used in studies on ecological footprint and sustainability, especially in calculating biocapacity. Given the importance of the modifiable areal unit problem (MAUP; i.e. the scale dependency of area-based information), a comprehensive understanding of how the changes of biocapacity across scales (i.e. the resolution of data) is pivotal for regional sustainable development. Here, we present case studies on the effect of spatial scales on the biocapacity estimated for two typical river basin and watershed in Northwest China. The analysis demonstrated that the area sizes of major land covers and subsequently biocapacity showed strong signals of scale dependency, with minor land covers in the region shrinking while major land covers expanding when using large-grain (low resolution) data. The relationship between land cover sizes and their change ratio across scales was shown to follow a logarithm function. The biocapacity estimated at 10×10 km resolution is 10% lower than the one estimated at 1×1 km resolution, casting doubts on many regional and global studies which often rely on coarse scale datasets. Our results not only suggest that fine-scale biocapacity estimates can be extrapolated from coarse-scale ones according to the specific scale-dependent patterns of land covers, but also serve as a reminder that conclusions of regional and global un-sustainability derived from low-resolution datasets could be a fallacy due to the MAUP.

1. Introduction

Since the concept of sustainable development was put forward (WCED, 1987), it has become an ideal development mode and a common policy goal. To date, many indicators have been developed to assess the status of sustainable development, such as the life cycle assessment (Robèrt et al., 2002), human development index by the UNDP (1990), barometer of sustainability (IUCN/IDRC, 1995), index of sustainable economic welfare (Daly and Cobb, 1989), environmental pressure indicator (EU, 1999), genuine progress indicator (Cobb et al., 1995), sustainable technology development (Weaver et al., 2000), environmental sustainability index (Siche et al., 2008) and ecological footprint (EF; Rees, 1992; Wackernagel and Rees, 1996). Among these large numbers of indicators of sustainable development, the EF methodology has gain popularity due to its compatibility with the data format commonly derived from economic and social surveys.

The EF for a particular population is defined as the total area of productive land and water ecosystems required to produce sufficient resources and assimilate wastes (Rees, 1992). Rees and Wackernagel (1994) further consider EF as the appropriated carrying capacity (i.e. human demand on nature) and biocapacity (BC) as the locally available carrying capacity of the ecosystem for generating resources and absorbing wastes. EF and BC, thus, represent the demand on and the supply from a regional ecosystem, respectively (Galli et al., 2007). As both EF and BC are measured in the same unit (the global hectare: gha), it is straightforward to calculate regional ecological budget as surplus and deficit (Rees, 1992). To this end, an ecological surplus (BC > EF) has been proposed as a minimum criterion for sustainability (Kitzes et al., 2009).

The EF framework, including both the concepts of EF and BC, are highly operable and easy to understand by the public and policy makers, with the data required accessible from government yearbooks. To date, EF has been applied at a variety of spatial scales, from municipality/provincial level (Solís-Guzmán et al., 2013) to national/global extents (Galli et al., 2012), covering all aspects of socioeconomic sectors, such as industry (Herva et al., 2012), education (Gottlieb et al., 2012), agriculture (Kissinger, 2013; Cerutti

Abbreviations: BC, Biocapacity; EF, Ecological Footprint; GIS, Geographic Information System; IDRC, The International Development Research Centre; IUCN, The World Conservation Union; JRW, Jinghe River Watershed; MAUP, Modifiable Areal Unit Problem; SRB, Shiyang River Basin; UNDP, United Nations Development Programme; WCED, World Commission on Environment and Development.

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0301-4797/$ – see front matter
http://dx.doi.org/10.1016/j.jenvman.2013.04.022

et al., 2013; Samuel-Fitwi et al., 2012), tourism (Castellani and Sala, 2012) and waste management (Herva and Roca, 2013).

As a continuously developing field the EF methodology has been widely criticized and mended. For instance, it has been considered a static indicator of weak sustainability as no dynamics and bounds are imposed on the level of ecosystem services and their demands. This has been partially solved by time series analysis and extrapolation. To project the future trend of regional sustainability, Haberl et al. (2001) calculated annual Austrian EF from 1926 to 1995. Senbel et al. (2003) examined the effects of consumption, ecological productivity and material efficiency on the ecological budget of North America over this century. Yue et al. (2006) used two quantitative indices (change rate and scissors difference) and depicted the long-term trend of EF and BC from 1991 to 2015 in the Gansu Province of China. Moore et al. (2012) used a Footprint Scenario Calculator to convert projected consumption and emission quantities and forecasted the trend of annual ecological budget up to 2050 (also see other examples from Niccolucci et al., 2012; Kuzyk, 2012; Vačkář, 2012).

The EF methodology has been rapidly developed in the last decade. To list a few, Bicknell et al. (1998) proposed an input–output framework for assessing the footprint of trading. Venetoulis and Talberth (2008) also improved the calculation of equivalence and yield factors – two weights assigned to each type of land cover for calculating the EF – by introducing the concept of net primary productivity into the EF framework. The calculation of EF has been standardized by the Global Footprint Network (2009). Siche et al. (2010) further combined energy analysis with ecological budget analysis and suggested to include low productivity land types in the calculation of biocapacity. Recently, Shao et al. (2012) proposed a modified exergetic indicator as a supplementary to conventional EF methodology.

As conventional EF methodology ignores management actions and policies, it only provides limited support to decision-making. The introduction of spatial features, with the help of the geographic information system (GIS), has largely released the EF methodology from this constraint (Mayer, 2008). For instance, to address the low accuracy and the lack of spatial heterogeneity of the conventional EF method, Yue et al. (2006, 2011) and Moran et al. (2009) introduced the remote sensing and GIS into the EF methodology, promoting the spatial analysis of EF and BC. We here focus on the scale dependency of BC when evaluated using GIS-based information and examine how such scale dependency affects the regional ecological budget and subsequently the fallacy of unsustainable development.

To calculate the biocapacity of a region, one first needs to estimate the available areas of biologically productive land and water. Specifically, this biologically productive area can be divided into six main categories (cropland, grazing land, fishing land, forest, built-up area and barren land; Chang and Xiong, 2005), and the sizes of these six land covers can then be either retracted from government agencies or increasingly calculated using remote sensing images with the aid of GIS (Wackernagel and Yount, 2000). However, in doing so, we often neglect an important issue that is associated with any spatial or area-based information – the scale dependency of spatial features (specifically here, the area sizes of different land covers). Evidently, the shape and size of different land covers are sensitive to the spatial scale (i.e. the resolution) of the maps as most landscape features are scale dependent and have self-similar, fractal structures (Mandelbrot, 1973). This scale dependency has been known in geography as the modifiable areal unit problem (MAUP; Openshaw, 1984) and is well recognized in spatial ecology (e.g. Kunin, 1998; Wu et al., 2000; Hui and McGeoch, 2008; Hui et al., 2006, 2010). Since the area-based information has been widely implemented for estimating the sizes of different land covers and therefore the BC (e.g. Hansson and Wackernagel, 1999; Wackernagel and Yount, 2000; Yue et al., 2006, 2011), it is important to assess how the BC estimated will be affected by the resolution of the available data and whether this scale dependency will change our perception on regional sustainability.

To this end, we chose two typical river basins in Northwest China (Jinghe River Watershed and Shiyang River Basin) and calculated the biocapacity at different spatial scales based on remote sensing data. This allowed us to further examine whether the conclusion of ecological deficit or surplus of the study areas depends on the resolution of the available data. In brief, we aim to capture the general patterns of this scale dependency of different land cover sizes and biocapacity, and further use the patterns captured to remedy the potential flawed conclusion of unsustainable development in many large-scale studies.

2. Materials and methods

2.1. Study areas

The Jinghe River Watershed (JRW; Fig. 1A) is a mountainous watershed located in the Midwest Loess Plateau (between 106°14′–108°42′E and 34°46′–37°19′N), covering an area of 44,983 km^2. The JRW has a typical temperate continental climate, with an annual average temperature of 8 °C and an annual precipitation of 350–600 mm. The main land categories are grassland (48%) and farmland (40%), with more than 80% of the northern watershed degraded severely from soil erosion. The Shiyang River Basin (SRB; Fig. 1B) is located in the transition zone of the Qinghai-Tibet Plateau to the Alashan Plateau (between 101°41′–104°16′E and 36°29′–39°27′N), covering an area of 41,600 km^2. The SRB has a temperate continental arid climate, with an annual average temperature of 7.2 °C and an annual precipitation of 60–610 mm. Most areas are covered by the barren land desert (48%). The nearest part of JRW and SRB are 22 km apart, and both areas have relatively equal size but distinct climates, topographies and vegetations (Liu and Wan, 2010; Zhao et al., 2011), ideal for comparing the scale dependences of BCs.

2.2. Data analysis and calculation

Following Rees (1992) and Rees and Wackernagel (1994), we calculated the biocapacity (BC) according to the available area of biologically productive land and water as follows:

$$BC = \sum_i A_i \times YF_i \times EQF_i \quad (1)$$

where A_i is the biologically productive area of land cover category i; YF_i is the yield factor of land category i and is calculated annually as the ratio of the local yield of a generic product to the global average yield of the same product (Zhang et al., 2001). The yield factor converts local biologically productive land into unites of global average productivity and thus facilitates comparisons across regions (Bastianoni et al., 2012). EQF_i represents the equivalence factor of land cover category i and is a scaling factor needed for converting a specific land use type into a universal unit of biologically productive area (gha) (Bastianoni et al., 2012). Equivalence factor is also calculated each year as the ratio of the global average productivity of a specific land type to the average productivity of all biologically productive land on the earth (Zhang et al., 2001). For JRW and SRB, the yield factors were estimated by comparing the average yield of the two watersheds with the global yield of different land covers. The equivalence factors were estimated using the data of the global yield of different land covers in specific years. The biocapacity of barren land was assigned to be zero in the calculation due to its extremely low productivity (i.e. the yield and equivalence factors of the barren land were zero; Table 1).