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REVIEW

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Double counting in ecosystem services valuation: causes and countermeasures

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Abstract Many studies have been conducted on ecosystem services and double counting is a frequent problem that causes uncertainty and poor reliability of estimating the value of ecosystem services. By referring to previous studies of ecosystem service valuation, our research has identified the basic causes of double counting. These include ambiguous definitions and inconsistent classifications of ecosystem services, poor understanding of ecosystem complexity, inadequate recognition of exclusiveness and complementarities of individual ecosystem services, spatio-temporal scale dependence of ecosystem services, and overlap and lack of cross-referencing between ecosystem service valuation methods. Measures for reducing double counting in ecosystem service valuation are proposed as follow: (1) identifying the spatio-temporal scales of ecosystem services; (2) valuing the final benefits obtained from ecosystem services; (3) establishing consistent classification systems for ecosystem services; and (4) selecting valuation methods appropriate for the study context.

Keywords Double counting · Ecosystem services · Valuation · Ecosystem function · Classification

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Introduction

Double counting is a term originally used in economics to refer to the erroneous practice of counting the value of a nation's goods and services more than once. The valuation of ecosystem services is also plagued by the double counting of goods and services provided by natural resources and the environment (Boyd and Banzhaf 2007; Fisher and Turner 2008). Ecosystem services refer to natural conditions and utility provided by ecosystems and ecological processes that sustain human life (Daily 1997). They are composed of interdependent components that are non-linear (Koch et al. 2009), which makes it impossible to separate each individual ecosystem service. The Millennium Ecosystem Assessment (MA 2005a), a UN-sponsored global ecosystem service valuation program, classified ecosystem services into *provisioning services* such as the production of food, timber, and fiber; *regulating services* such as climate regulation, flood regulation, and water quality control; *cultural services* such as recreational, aesthetic, and spiritual benefits; and *supporting services* such as soil formation, photosynthesis, and nutrient cycling. It is known that there is overlap between individual ecosystem services in the widely used MA classification system, which inevitably leads to double counting. For example, surface water flow is a regulation service, water quality improvement by infiltration through soil is a supporting service, and potable water supply is a provision service. Simply aggregating these services will lead to double counting because water flow regulation and water treatment by soil help to provide the same final product of potable water (Fig 1).

Costanza et al. (1997) classified global ecosystem services into 17 categories and assessed the total monetary value of global ecosystem services in the range of \$18–\$61 trillion, with an approximate average of \$38 trillion. Daily (1997) presented guidelines for ecosystem service valuation including consideration of competing approaches, ethical biases, and social decisions. The

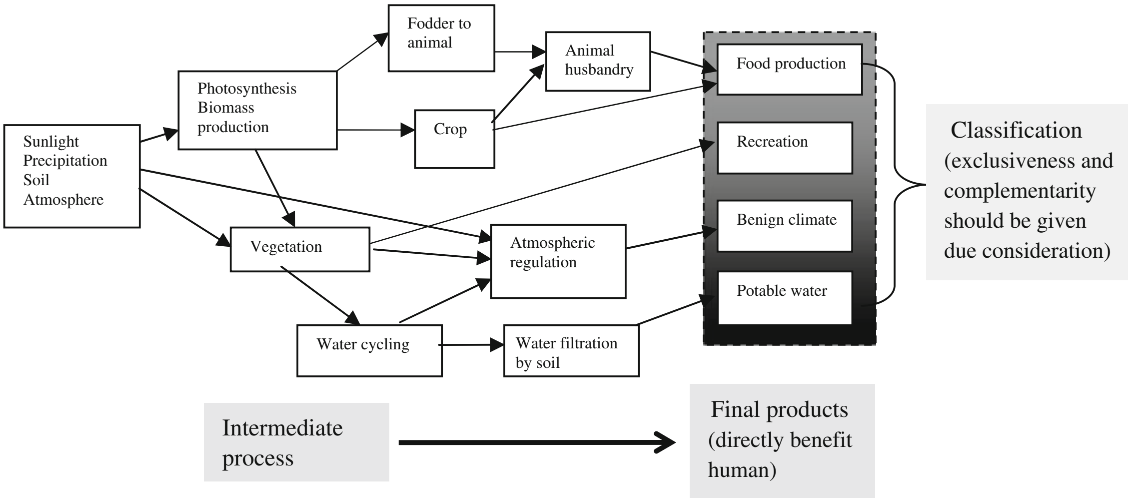

Fig. 1 Diagrams of delivering mechanisms for some ecosystem services. Services with *dark grey* backgrounds on the right side are final services; *boxes* on the left sides are the intermediate process or products. The final products are the results of web-like intercorrelations of different intermediate process or products. The criterion of final product is decided by human at which stage the product was reaped

studies of Costanza et al. and Daily, together with the special issue on "Values of Ecosystem Services" published by *Ecological Economics* in 1998 (Alexander et al. 1998) led to an upsurge of ecosystem services valuation. Researchers have attempted to value ecosystem services from different perspectives, such as biodiversity (Lewandrowski et al. 1999), urban ecosystems (Bolund and Hunhammar 1999; Kreuter et al. 2001), or wetland ecosystems (Woodward and Wui 2001). Studies have been made on spatial scale-dependency (Konarska et al. 2002), and multiple valuation methods have been attempted (Li et al. 2006; Ingraham and Foster 2008). Few of these studies have addressed the issue of double counting. As mentioned by some researchers (Serafy 1998; de Groot et al. 2002; Turner et al. 2003; Hein et al. 2006; Boyd and Banzhaf 2007), double counting has not yet received due attention as a major cause of inaccuracy and unreliability of ecosystem service valuation. There are large differences in valuations even for the same ecosystems due to double counting. For example, there were differences of an order of magnitude for global ecosystem services valuations between those of Costanza et al. (1997) and Pimentel et al. (1997). To date, there has been little published to specifically address problems caused by double counting. This paper attempts to probe the fundamental causes of double counting in ecosystem service valuation and seeks to provide some useful countermeasures.

Fundamental causes of double counting

There are many causes of double counting in ecosystem service valuation. We analyzed the fundamental causes of double counting from perspectives of ecosystem service definition, exclusiveness and complementarity of ecosystem services, complexity of ecosystems, spatio-temporal scale dependence of ecosystem services, ecosystem service classification, and ecosystem service valuation methods.

Ambiguity in definition of ecosystem services

The most frequent confusion is between ecosystem *functions* and ecosystem *services* and the distinction between them has not been given enough attention. Ecosystem functions are the intermediate processes that are necessary and conducive to the end product of ecosystem service. As an intrinsic property of ecosystems, ecosystem *functions* refer to the physical, chemical, and biological processes that contribute to the maintenance of an ecosystem—including material circulation, energy flow, information connections, and their dynamic evolution (Odum 1971; Wallace 2007). In contrast, ecosystem *services* refer to the services provided by ecosystem conditions and processes that are direct benefit to human beings. Definitions of ecosystem services tend to be broad, and this is likely to be the cause of some confusion. They are the benefits that people obtain from ecosystems, and serve as the end-products for human well-being (Wallace 2007). Confusion of ecosystem functions and services leads to errors in ecosystem services valuation. It is obvious that simply aggregating ecosystem services with ecosystem functions to calculate overall ecological values tends to result in double counting. To address this, Wallace (2007) divided ecosystem services into direct (final) and indirect (intermediate) services, and recommended that delineating direct and indirect ecosystem services reduces the risk of double counting. For example, food provision is a final service whereas the pollination of food crops is an intermediate service, so that the value of the pollination should be incorporated in the value of food production. Boyd and Banzhaf (2007) narrowed the scope of ecosystem services valuation by counting only the natural elements that are beneficial and directly consumed by humans, while excluding other natural elements. For example, in assessing wetland ecosystem services, potable water is regarded as one of the ecosystem services since it can provide drinking water for human consumption. In contrast, water cycling and soil filtration are ecological functions that provide this ecosystem service. In this case, only the value of the potable water should be included in the calculation of ecosystem value, and the values of water cycling and soil filtering should be excluded (Fig. 1). Another example is atmospheric regulation (and its related services and functions) and climate regulation. Since the former has some impact on the latter, double counting would occur if they are estimated separately and aggregated (Fig. 1). The exclusion of intermediate functional services in ecosystem service valuation does not indicate that they have no value but that their values are realized through the value of the final ecosystem services.

Complexity of ecosystems

Ecosystems are characterized by inherent complex interrelations and are resilient to external disturbances (Carpenter et al. 2006). They are characterized by biologically, socially, and physically heterogeneous components interacting at several spatio-temporal scales, incorporating organizational connectivity or structure and contingency through time (Cadenasso et al. 2006). For example, hydrological regulation, nutrient cycling, and climatic regulation are interrelated with each other and affected by a common factor of vegetation coverage (Turner et al. 2003). Therefore, ecosystem complexity results from their intrinsic features including multiple structures of different components, dynamic variations and interrelations between biological entities, spatio-temporal variation of biological entities, and interactions with external environments. Natural communities per se of living organisms are complex systems with internal structure and inherent organization (Azovsky 2009).

As ecosystems provide multiple complex services, we need to disaggregate total ecosystem services into independent services, value each individual service, and then sum them to determine the total monetary value of the total ecosystem services. However, it is not possible to disaggregate total ecosystem service because the individual component services do not strictly correspond to individual ecosystem functions, and most of the individual ecosystem services are dependent upon or overlap with others (Odum 1986; Munasinghe 1994; Costanza et al. 1997; Daily 1997). In addition, the mechanism of natural ecosystem processes and their intrinsic interactions have not yet been fully understood, which composes an underlying cause of double counting. Barbier (1994) argued that the function of nutrient retention in tropical wetlands was essential for maintaining biodiversity, but double counting would occur if the value of nutrient retention and that of biodiversity conservation are measured individually and added together. Double counting also occurs in valuation of wetland ecosystem services when simply adding the value of recreation and aesthetics service with those of waste water treatment and water storage (Turner et al. 2003). Ecosystems are so complex that it seems more meaningful to study the change of individual ecosystem service values due to ecosystem changes than to quantify the absolute value of total ecosystem services (Yang et al. 2006).

Ecosystem complexity is also caused by the complicated causalities within and between ecosystem services. For example, regulating services are the benefits obtained from the regulation of ecosystem processes, including regulation of air quality, climate, water, and soil erosion, water purification and waste treatment, disease regulation, pollination, natural hazard control et al. (MA 2005a). Provisioning services are the products obtained from ecosystems, including food, fiber, fuel, genetic resources, biochemicals, ornamental resources, fresh water, etc. (MA 2005a). The final services, and, in most cases, the provision services (or end services), are the result of regulation services (intermediate service). For example, the ecosystem service of food production is the result of regulating services of soil formation, primary production, nutrient cycling, and water cycling, even though the regulating services are not directly usable to humankind. These complex causal relations blur the boundaries of different components of ecosystem values, e.g., the interaction of direct use and indirect use of ecosystems. It is an essential prerequisite to clarify the causalities before assessing total ecosystem service value. In a coastal wetland ecosystem, for example, nutrients captured from watercourses in the upper reaches of rivers can help the growth of fish and shrimps in wetland areas. Double counting can readily occur if the value of nutrient interception is added to that of fish and shrimp production, as value of nutrient interception has already been included in the direct value of fish and shrimp production.

Spatio-temporal dependence of ecosystem services

Scales of ecosystems, scales of socioeconomic systems, and scale of interests of stakeholders in relation to ecosystem services are three key issues in valuating the scale dependence of ecosystem services (Hein et al. 2006). Scales refer to the physical dimensions, in space or time, of phenomena or observations (O'Neill and King 1998). Ecosystem services derive from a range of spatio-temporal scales, varying from the short-term site level (e.g., amenity services) to the long-term global level (e.g., carbon sequestration) (Turner et al. 2000; Limburg et al. 2002), and the timeframe and geographic scales have strong impacts on the value different stakeholders attach to services. Certain ecosystem services may be best considered at specific scales (MA 2005b), e.g., water

Table 1 Spatial scales for ecosystem services

Spatial scale	Scope (km^2)	Examples of leading ecosystem services at different scales
Global	$>10^6$	CO_2, N, P circulation, climate change regulation
Regional	10^4–10^6	Controlling flood, protecting ground water, nature reserve
Landscape	1–10^4	Soil conservation, decomposing pollutants, species habitat
Ecosystem	<1	Water conservation, biological nitrogen fixation

regulation is best considered at a watershed scale, while carbon sequestration can be considered on a national or global scale (Table 1). To date, there are relatively few studies of the spatial scales of ecosystem services (MA 2003; Turner et al. 2003). Research on temporal scales of ecosystem service is even scarcer, even though sometimes it is necessary, e.g., impacts on biodiversity today may have effects on future generations (Hein et al. 2006). Scales and stakeholders are often interrelated because the scale at which an ecosystem service is supplied determines which stakeholders accrue benefits (Vermeulen and Koziell 2002). For example, at large spatio-temporal scale, CO_2 cycling is relevant to stakeholders at the level of states or countries, while at seasonal and site scales the provision of some biological products is the concern of individual humans as stakeholders.

Ecosystem services result from different ecosystem processes at various scales, such as competition between individual plants at habitat scale, susceptibility to fire, plant disease, insect pests at regional scale, and climate and typological processes at global scale (Kremen 2005). The interrelations between various ecological processes on different scales are very complex. For example, large-scale ecological processes of biodiversity, transfer of pollutants, and greenhouse effects arise from small-scale environment variation (Levin 1992). Likewise, small-scale ecological processes such as nutrient cycling and soil formation are affected by large-scale variations such as global climatic change and fluctuations of ocean currents (Lü and Fu 2001). There are various trade-offs or synergies of ecosystem services across different scales. Over-emphasis of ecosystem services at a certain scale will cause losses at other scales. For example, excessive exploitation of food production services from grasslands will cause the loss of sand stabilization functions, and lead to desertification. The extent that ecosystem services play their roles is also strongly scale-dependent. For example, timber logging affects water resource conservation only slightly at the site scale but the effects will be serious at larger scales.

The scale dependency of ecosystem services presents decision-makers with a new problem—how to balance the interest of different stakeholders at different scales. Stakeholders at different scales view ecosystem services differently due to their cultural background and the impacts of ecosystem service on their income and the environment (Hein et al. 2006). Comprehensive analysis of the interrelations between scale and stakeholder helps to solve conflicts of interest of different stakeholder (Tacconi 2000) and can provide scientific basis for multiscale decision-making on ecosystem management (Rodriguez et al. 2006).

Exclusiveness and complementarity of ecosystem services

Ecosystem services are often public goods that can be enjoyed by the general population without affecting others' enjoyment. For example, no matter how many people enjoy an aesthetic view, others can enjoy it too. Exclusiveness and complementarity are always used to describe the 'public/private' traits of social capital. Externality and internality, a pair of opposing terms frequently used in the field of economics, can best describe exclusiveness and complementarity. Ecosystem services of complementary always have the characters of externality, i.e., such ecosystem services can be shared by multiple stakeholders without diminishing themselves. People living downstream experience a negative externality if a stream is polluted from agricultural activities upstream. Exclusiveness means ecosystem services enjoyed by some people will preclude other people's enjoyment of them, i.e., such ecosystem services have the characteristic of internality. Exclusiveness is a necessity for establishing property rights and markets for ecosystem services (Randall 1987). The openness of ecosystems indicates that most ecosystem services and functions are external and free of charge, but there are small proportions of ecosystem services, especially in the provision services, that can be readily captured for the market, and therefore are exclusive, e.g., food, fiber, and natural medicine provision.

There are some ecosystem services that share properties of both exclusiveness and complementarity, i.e., they are semi-exclusive and semi-complementary, such as tourism services of public recreation areas. Dai et al. (2005) divides ecosystem services into four categories according to their marketability: (1) ecosystem services whose value can be directly realized in markets and feature 'private goods'. Services in this category are exclusive, e.g., foods, raw material, and ornaments; (2) ecosystem services whose value can be indirectly realized in markets, and feature 'quasi-public goods', such as water purification and disaster prevention; (3) ecosystem services are difficult to express in economic systems, such as atmospheric regulation or nutrient cycling; and (4) ecosystem services that are independent of economic systems, such as the customs, religion, and psychology of humans. Among the four categories of ecosystem services, only the exclusive category (1) can be fully commoditized, while other categories are non-exclusive and cannot be fully represented in the market. Aggregation of the exclusive ecosystem services of category (1) with services of the categories (2–4) will inevitably lead to double counting. For example, in forest ecosystems, timber can be traded as goods and is therefore exclusive, while other ecological values of forest ecosystem such as water purification, soil retention, atmospheric regulation, and recreation are non-exclusive. If the timber is harvested and sold as goods, other ecological services of the forest will no longer exist. Counting the value of timber production with water purification, soil retention, and atmospheric regulation will result in double counting.

Disputes exist about whether to count some complementary ecosystem services. For example, carbon sequestration and oxygen release are two complementary services that can be taken as indicators for carbon and oxygen balance regulation. Ouyang et al. (1999) argued that carbon storage and oxygen release have their own values, and they should be summed to obtain the value of maintaining carbon and oxygen balance. However, Xue (1997) argues that services of carbon storage and oxygen release belong to the same ecological process and the value of oxygen release should be excluded, while the sole value of carbon storage should be taken as the surrogate for the value of maintaining carbon and oxygen balance.

Trade-off and synergy are two opposing phenomena closely related to exclusiveness and complementarity. Trade-off is frequent in exclusive ecosystem services, while synergy always exists in complementary services. Both trade-offs and synergies are the result of common drivers behind the simultaneous ecosystem services or interactions across different services (Hughes et al. 2007). Without knowledge of the relationship between ecosystem services, we are at risk of incurring unwanted trade-offs and neglecting unintended synergies (Bennett et al. 2009). In order to analyze the interactions among multiple ecosystem services, Raudsepp-Hearne et al. (2010) designed a valuation scheme—an ecosystem service bundle of sets of ecosystem services that repeatedly appear together across space or time. This scheme captures different ecosystem services interactions, and is "distinct from inventories of ecosystem services that is added up to obtain a total quantity of services" (Raudsepp-Hearne et al. 2010), as "adding the services within a bundle would double count ecosystem services that interact". Swallow et al. (2009) extended the theory of trade-off and synergy by further dividing synergy into synergy and environment. To illustrate their argument, Swallow et al. (2009) provided an example of high grain production with high sediment yield that indicates trade-offs between provisioning and regulating services. High grain production with low sediment yield indicates synergies between economic development, whereas low production with high sediment yield indicate an environment–poverty trap. He further explained that synergy demonstrated a virtuous or win–win cycle, while environment–poverty trap demonstrated a vicious or lose–lose cycle.

Inconsistency of ecosystem service classification systems

Ecosystem service classifications are made for different purposes (Fisher and Turner 2008). Inconsistency in ecosystem service classification systems is another major cause of double counting. There are various classification systems in the literature for ecosystem services (Table 2). For instance, as many as 15 ecological services are identified for forest ecosystems: air purification, water conservation, climate regulation, absorption of waste, soil conservation, pollination, disease and pest control, nutrient cycling, seed dissemination, biodiversity conservation, provisioning of key agricultural and industrial production, ultraviolet ray protection, wind control, forest culture, and forest recreation (Yu et al. 2005; Sohngen and Brown 2006; Matero and Saastamoinen 2007; Koellner et al. 2008; Watson 2008; Bond et al. 2009; Jim and Chen 2009; Lara et al. 2009; Patterson and Coelho 2009). Based on links between human welfare and services provided by ecosystems, MA (2005a) divides ecosystem services into supporting, regulating, provisioning, and cultural service (Table 2). This classification system is more a heuristic tool than a practical one (Fisher et al. 2008). For example, in MA (2005a), nutrient cycling is a supporting service, water flow regulation is a regulating service, and recreation is a cultural service. There are some overlaps in that the first two services provide the same service in the form of usable water, and the third one turns the usable water into a direct benefit (recreation on a clean, navigable river). Double counting will occur if these services are individually valued and summed.

Turner et al. (1994) categorized ecosystem services into two major groups: use value and non-use value (Table 2). Use value was further divided into direct use value, indirect use value, and option value. Non-use value was divided into bequest value, existence value, and non-demand ecosystem value (Table 2). Hawkins (2003) established a classification system analogous to that of Turner et al. (1994). The difference is that the optional value is as a use value while Hawkins categorized it as non-use value. The other difference is that the service of 'non-demand value' in the non-use category of Turner et al.'s classification system is absent from Hawkins' classification system (Table 2). Hein et al. (2006) argued that it is not necessary to include non-demand and bequest services. The classification systems of Hawkins and Turner et al. have some overlaps. For example, the direct use value and indirect use value are existence value to some extent, while the differences among option value, bequest value, direct use value and indirect use value are mainly due to the temporal scale. Overlaps also exist between option value and bequest value, which was claimed by Ouyang and Wang (2000) as the same value.

Table 2 Major classification systems for ecosystem service valuation

Source	Ecosystem service types					
Costanza et al. (1997)	**17 types** Gas regulation, climate regulation, disturbance regulation, water regulation, water supply, erosion control, soil formation, nutrient cycling, waste treatment, pollination, biological control, habitat/refuges food production, raw material, genetic resources, recreation, cultural					
de Groot et al (2002)	**Regulation functions** Gas regulation; climate regulation; disturbance prevention; water regulation; water supply; soil retention; soil formation; nutrient regulation; waste treatment; pollination; biological control	**Habitat functions** Refuge function; nursery function	**Production functions** Food; raw material; genetic resources; medicinal resources; ornamental resources	**Information functions** Aesthetic information; recreation; cultural and artistic information; spiritual and historic information; science and education		
MA (2005)	**Regulation services** Air quality regulation; climate regulation; water regulation; erosion regulation; water purification and waste treatment; disease regulation; pollination; natural hazard regulation	**Supporting services** Net primary production; soil formation; nutrient cycling; water cycling; refugia providing	**Provisioning services** Food; fiber; genetic resources; biochemicals, natural medicines pharmaceuticals; fresh water	**Cultural services** Spiritual/religious value; aesthetic values; recreation/ecotourism; inspiration/education		
Turner et al (1994)	**Non use value** **Bequest value** Leave a legacy for future generations. such as vegetation plantation for future use	**Existence values** Knowing the environment is out there; knowledge of the existence of marine; biodiversity	**Non-demand ecosystem service** The value of life itself or intrinsic value	**Use value** **Indirect use value** Mainly ecosystem function value: habitat provision; nutrient and soil retention; regulation of water quality and quantity	**Direct use value** Consumptive and non-consumptive: recreational use of parks, forests; grazing; agriculture; fishery; gene harvest	**Option value** Retaining the option for future use: future visits to sites of interest; future access to genetic material
Hawkins (2003) [a]	**Existence value** Biodiversity, water conservation value; soil conservation;			**Indirect use value** Biogeochemical cycling and water cycling; species/genetic diversity; soil fertility; atmosphere equilibrium/stability	**Direct use value** Food; pharmaceutical; raw material for agriculture and industry; landscape recreation	**Option value** Water conservation; air purification; recreation etc.

[a]Hein et al. formulated a classification similar to Hawkins, except that Hein named the existence value as non-use value

Table 3 List of ecosystem service valuations at different scales and of different classification system, with difference existing for same ecosystem due to double counting

Sources	Ecosystems valued	Classification system adopted	Spatial and temporal scales	Valuation methods	Values calculated	Is there double counting? What is it?
Costanza et al. (1997)	19 types of marine and terrestrial ecosystems	17 groups: gas regulation, climate regulation; disturbance regulation; water regulation; water supply; erosion control; soil formation; nutrient cycling; waste treatment; pollination; biological control; habitat refugia; food production; raw material; genetic resources; recreation; culture	Global	Synthesized method: energy analysis method Willingness to pay contingent value method stated preference method supply-demand analysis Market value method Partial equilibrium analysis	US$33 trillion per year	Yes. Nutrient cycling overlapped with waste treatment; recreation overlapped with culture.
Brown and Henry (1989)	Wild elephant resort	Gross tourism viewing value	South Africa	Travel cost method	US$25-30 million	Yes Legacy use overlapped with option value
Tobias and Mendelsohn (1991)	Tropical rain forest	Ecotourism	Monteverde cloud researve, Costa Rica	Travel cost method	US$97500~ US$116200	No
Gren et al. (1995)	Forest/grasslands/ permanent water	Resources, nutrient sinks, recreation	Danube floodplains, Romania	Transferring results obtained in other studies	US$866 million	Yes Resources and nutrient sinks somewhat overlapped
Bolund and Hunhammar (1999)	Street tree; lawn/parks urban forest; cultivated land; wetland; stream/lake; sea	Air filtering; micro climate regulation; noise reduction; rainwater drainage; sewage treatment; recreation; cultural values	Stockholm, Sweden	Cost-benefit analysis Shadow project method hedonism method substitute value	No total value	Yes. Air filtering overlapped with micro climate regulation; rain water drainage overlap with sewage treatment; recreation overlap with cultural values
He et al. (2005)	22 types	NPP; nutrition store and transfer; water conservation; soil conservation; carbon fixation and oxygen release	China (terrestial)	Shadow project method; substitute value; market value method; carbon tax method; Stated preference method	US$ 1132.1 billion	Yes. Soil conservation overlap with nutrition store and transfer
Pan et al. (2004)	Tropical forest; temporal forest; grass; red wood; swamp; desert; tandra; ice/rock; farmland	Refer to Costanza et al.	China (terrestial)	Remote sensing data plus Costanza et al.'s ecosystem service value coefficient	US$ 795.6 billion	Yes As Costanza et al.'s

Wallace (2007) argued that the classification of MA mixes process (means) for achieving services and the services themselves (ends) within the same classification category, i.e., many individual services in supporting and regulating functions are the means to provide the production function and culture/tourism functions. For instance, nutrient and water cycling as supporting functions are not services pursued by humans, but are merely means to secure fresh water and food. Pollination of flowers of crops, as a regulating function, is also not of direct interest to humans. Its real interest for human is the resultant food production. In response to Wallace's argument, Costanza (2008) noted that the basic premise Wallace took was flawed and suffered from a gross oversimplification of a complex reality, e.g., the "benefit" is too narrow to represent the public service. He provided two useful ways to improve Wallace's classification, i.e., classification according to spatial characteristics and classification according to "excludability/rivalness" status.

Hein et al. (2006) also provide insight into the classification system of MA (2005a). In contrast with Wallace, Hein et al. (2006) argue that the supporting functions are reflected in the other three functions—provisioning, regulating, and cultural. In the case of the regulating service, Hein et al. (2006) argues that regulating values should only be included in the overall ecosystem services values under two conditions. Firstly, the individual regulating value has some impact outside the ecosystem to be valued. Under this condition, it is the spatial configuration and the interactions with ecosystems of society outside the studied system that determine the value of the service. For example, if an ecosystem provides support to a swarm of bees that plays a significant role in pollination of crops in the surrounding fields, the value of the swarm should be included in the overall ecosystem service value. Secondly, the regulating services provide a direct benefit to the local people. For example, if noise and dust have caused some negative impact on the life of people living beside a highway, the value of green belts along the highway that helps reduce noise and dust should be included.

Other researchers (de Groot et al. 2002; Groffman et al. 2004; Anielski and Wilson 2005; Eamus et al. 2005) also tried to classify ecosystem services into different categories, but failed to distinguish ecosystem services from ecosystem functions. As a consequence, inconsistent classification systems lead to an overlap of individual ecosystem services, which makes the valuation process susceptible to double counting. For example, there were differences of an order of magnitude in the values for global ecosystem services between the results of Costanza et al. (1997) and those of Pimentel et al. (1997). Costanza et al. (1997) categorized an ecosystem service into 17 types, which have some risks of overlap, e.g., water regulation versus water supply, recreation versus culture, erosion control and sediment retention versus soil formation versus nutrient cycling, waste treatment versus nutrient cycling, etc. Compared to Costanza et al. (1997), Pimentel et al. (1997) categorized an ecosystem service into 13 types, which also have some risks of double counting, e.g., biomass and recycling of organic wastes versus bioremediation of chemical pollution, perennial cereal grains versus harvest of food, and pharmaceuticals from the wild.

The differences of ecosystem service classification stem from the fact that every classification scheme is founded upon a specific context (Fisher and Turner 2008). The basic theory that different researchers use for their classification of ecosystem services also causes diversity in classification schemes. For example, the classification by MA (2005a) was centered on making the link between human welfare and services provided by ecosystems. Boyd and Banzhaf (2007) established their classification for the purpose of devising an accounting system for ecosystem services, which they defined as the directly consumed ecological components of ecosystems. Wallace (2007) focused on managing landscapes and ecological processes to deliver ecosystem services. Fisher and Turner (2008) thought that ecosystem services are the aspects of ecosystems that are utilized (either actively or passively) to produce human well-being. They established their ecosystem service classification, which mainly answers questions such as: how do ecosystem services deliver human welfare?; where are the benefits realized?; and how do their value changes across the landscape in regard to different future scenarios.

Significant differences exist among the terrestrial ecosystem services valuation for China due to inconsistent classification systems (Pan et al. 2004; He et al. 2005) (Table 3). Based on remote sensing, climatic and geographical data, He et al. (2005) calculated terrestrial ecosystem services of China in terms of five classes, i.e., solar energy fixed and food production, nutrient recycling, water supply and regulation, erosion control and sediment retention, and gas regulation. Pan et al. (2004) valued the terrestrial ecosystem service for China based on the classification and value coefficients of Costanza et al. (1997), which, as mentioned above, has a high risk of overlap.

Overlap and no cross-referencing of valuation methods

The complex traits and possible overlap of different valuation methods of ecosystem services are not negligible in terms of double counting. Ecosystem service valuation methods can be broadly categorized into marketable service methods, stated preference (hypothesized market) methods, and revealed preference (surrogate market) methods (Barbier 2007). Certain ecosystem services can be valued by diverse valuation methods, e.g., water supply can be valued by direct market, replacement cost, avoidance cost, travel cost, hedonic pricing, and contingent valuation methods (Hawkins 2003). Recreation services can be valued by direct market, contingent valuation, travel cost, and hedonic pricing methods (Hawkins 2003). The application of different methods engenders the risk of double counting in ecosystem service valuation processes. Take the three frequently used travel cost, hedonic pricing, and contingent valuation methods as an example. As a major 'revealed preference' method, the travel cost method has the basic premise that travel cost can represent the 'price' or value of the scenery, i.e., people's 'willingness to pay' is regarded as a token of the price of the scenery. The hedonic pricing method, as another 'revealed preference' method, takes hedonism as a symbol of the price of real estate or scenery. These two methods overlap in terms of 'amenity'. McConnell (1990) provides an example of double counting by travel cost and hedonic pricing: a lake is less attractive for fishing and swimming due to pollution, and the influence of pollution can also be found from the lowering of house prices near the lake and from reduced recreation use of the lake. Both the travel cost method of recreation and the hedonic pricing method of housing can produce measures of pollution damage. If the value of pollution mitigation is assessed by aggregation of the valuations from the housing market and the varying of demand for recreation then double counting will occur. Overlap also exists between the travel cost and contingent valuation methods in that both methods obtain prices of some unmarketable ecosystem service by asking people their willingness to pay. The application of these methods needs to be handled with great caution to avoid double counting.

Double counting arises not only across different valuation methods but also within certain individual methods. We will take the contingent valuation method and production function analysis method as a case study to elaborate double counting within an individual valuation method. As a survey-based 'stated preference' method, the contingent valuation method was primarily developed by neo-classical economists (Veisten 2007). Critics argue that in the process of contingent valuation, accepting 'willingness to pay' motivated by altruism leads to double counting in the cost-benefit analyses of public good provision service. Milgrom (1993) argued that altruism-motivated monetary values should not be counted in contingent valuation. Johansson (1992) demonstrated that double counting is closely related to the specification of the contingent valuation model and he argued that (non-paternalistic) altruist value can be dropped from the cost-benefit calculation since the altruist household 'values both benefits and costs of other households'. In addition, as altruism is the major motivation for 'willingness to pay' in the contingent valuation method, double counting can be avoided on condition that it is directed towards future generations or the environment itself, no matter paternalistic or non-paternalistic.

The production function method can be used to value the contribution an ecosystem service makes to produce some marketable services such as drinking water or food (Barbier 2007). This analytical method generally applies the cause–effect relationships between ecosystem services and the output level of the marketable commodity (Chee 2004). Production function analysis has been applied to valuing ecosystem services such as the environmental functions of tropical wetlands (Barbier 1994) and the groundwater recharge function of wetlands on agricultural production (Acharya 2000). The main limitation of this method is the lack of understanding of cause–effect linkages between ecosystem service and the commodity (Daily et al. 2000). As we have argued, ecosystems are complex, dynamic systems whose components often interact in nonlinear ways across spatio-temporal scales. As the production function approach focuses on the relationship between ecosystem service and marketable goods, the inherent interconnectivity and interdependencies of ecosystem services per se make this method prone to double counting. In addition, the production of various ecosystem commodities at different stages tends to cause overlaps and result in double counting (Fig. 1).

The fundamental causes of double counting in ecosystem service are not independent, but rather are interdependent. The complexity of ecosystems, inherent properties of ecosystem services, scale dependence of ecosystem services, inconsistent classification systems, and the overlap with no cross-referencing of ecosystem service valuation methods constitutes a paramount challenge for reducing double counting. There is some feedback between the ecosystem per se, inherent properties of ecosystem services, scale dependency of ecosystem services, and valuation methods of ecosystem services. It is necessary to disentangle these complicated interrelations and feedback to get a thorough understanding of the causes of double counting.

How to reduce double counting

Based on the above discussion and the outline of the fundamental causes of double counting, we present measures to reduce them in ecosystem service valuation as follows:

Identifying the spatio-temporal scales of ecosystem services

As the dominant ecosystem services vary with different spatio-temporal scales (Hein et al. 2006), identifying the scale of the studied ecosystem is a crucial step in ecosystem service valuation. In general, the large-size, long-term ecological services and functions constrain or control the small-size, periodical ecosystem services and functions (Limburg et al. 2002), with the former being the result of the combined effect of the latter (Levin 1992). Besides, human cognitive perceptions also vary at different spatio-temporal scales (Hein et al. 2006; Martin-Lopez et al. 2009). For example, at the small scale, direct, short-term benefits from consumption of products is preferred, while at large scales ecosystem services of conservation are preferred. In addition, ecosystem services and functions at various spatial scales promote or constrain each other. Even for ecosystems of the same type, their services and functions can vary greatly depending on their sizes, structures, and spatial overlaps with other types of ecosystems (Hein et al. 2006). The selection of the ecosystem service scale depends on the purpose of research and data availability, and is always conducted through an empirical analysis and decision-making processes (Zhang et al. 2007). The criteria for choosing an appropriate valuation scale include: (1) key ecological feedbacks are embodied within each individual ecosystem and (2) key correlations between different ecosystems are embodied (Zhang et al. 2007). The dominate ecosystem services need to be defined according to the selected scale. For example, ecosystem services of a bee hive include honey production and pollination. If the scale were extended to a whole orchard, then the service of pollination would be replaced by fruit production service, as pollination service has been included in the value of fruit production.

Selecting final benefits by disentangling the mechanism of ecosystem services

Economists avoid double counting by only including the value of final goods in GDP, i.e., goods that have been purchased for final use by a consumer without an immediate intent of resale or further processing. Compared to conventional economics, the valuation of ecosystems is fraught with complicated interrelations (Carpenter et al. 2006), and disentangling them is difficult. Only by disentangling the interactions and overlaps between ecosystem functions and services through in-depth probing of the structure, process, function, and mechanism of ecosystems from the perspective of basic ecology, can double counting be reduced. In calculating the value of ecosystem services, only the final ecosystem services should be included in the value of the total ecosystem service, and intermediate services be excluded. The criterion of the end point of the ecosystem service chain is human benefit (Wallace 2007). For example, provision of potable water is a major service of forest ecosystems, which can be produced through multiple stages: rainfall, vegetation control, water cycling, and water filtration by soil under vegetation canopy (Fig. 1). In a valuation of ecosystem service, only the final product—potable water—should be included, while the intermediate process should be omitted.

Establishing consistent classification systems for ecosystem services

Ecosystems and their services are integral. Within each category of ecosystem service there are interdependent and causal relations or overlaps. It is inappropriate to classify ecosystem services in this sense (Odum 1986; Munasinghe 1994; Costanza et al. 1997; Daily 1997). However, classification is necessary to measure the total value of ecosystem services. In order to avoid double counting, a detailed and consistent classification system of ecosystem services needs to be formulated to minimize the overlap of different categories. The classification system should be carried out with the final ecosystem services. There are two problems that need to be emphasized. Firstly, ecosystem structure and composition should be taken as indicators for ecosystem services rather than ecosystem processes in that the former are better surrogates for the latter rather than vice versa (Wallace 2007). The reasons are threefold: (1) there is poorer understanding of the full range of ecosystem processes than ecosystem elements, e.g., biodiversity; structure, and composition of ecosystems are more readily observed and measured than pollination regulation. (2) We can only manage limited ecosystem processes concerning ecosystem service—placing more stress on ecosystem processes will cause neglect of some concrete indicators such as species, or genotypes due to their low relevance to ecosystem process. (3) From the perspective of human well-being, tangible benefits, such as food, water, and property are preferred over ecosystem processes such as nitrogen of carbon cycling. Secondly, the point at which ecosystem services arise should be decided. Ecosystem services are benefits people obtain from ecosystems while benefits are always linked by ecological processes. Even the same ecological process can produce different benefits depending on the stage at which we reap the service. Take a food production chain, i.e., photosynthesis–cereal crop (fodder)–animal (meat) as an example (Fig. 1). There is a trade-off between cereal crop and meat depending on the stage taken by different stakeholders. It is important to decide the endpoint under a specific context—otherwise double counting may occur with different services at different stages and the valuation process will lose its logical foundation.

Some productive services of ecosystems can be traded in the market and are exclusive to each other (Randall 1987). Even within the regulating and supporting functions of ecosystem services, there are some opposing services, e.g., the draining of surface water and replenishing of ground water. Ecosystem services have both characteristics of externality (complementarity) and internality (exclusiveness). As major sources of double counting, contradictory and exclusive ecosystem services should be strictly differentiated in the classification in advance of the valuation. The evolution and overlap of ecosystem services across different scales need to be highlighted in order to avoid double counting, and the leading ecosystem services at different scales should be identified in assessing ecosystem services.

Selecting appropriate valuation methods with consideration of context

Humans estimate ecosystem services including direct use, indirect use, and non-use values (Hawkins 2003). The values of the services can be quantified using the market method, revealed preference method, and the

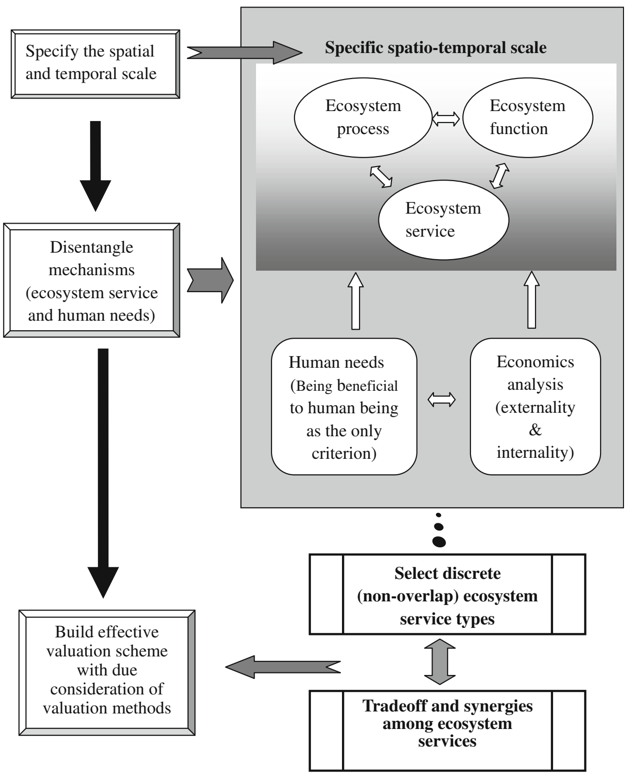

Fig. 2 Framework for reducing double-counting in ecosystem service valuation. In the *dark grey box* is the basic ecosystem mechanism, i.e., structure, process, and function, which constitutes the physical components for ecosystem services. The *light grey* area contains two prerequisites for ecosystem services, i.e., human needs and economic principle. The ecosystem mechanism, human needs, and economic principles composed the ecological and human driving factors for ecosystem services. An effective valuation scheme also needs to fully consider the overlap, trade-off, and synergy between different ecosystem services and select an effective valuation method for specific ecosystem service

stated preference method (Barbier 2007). Direct market pricing can be used for those marketable ecosystem services that are mostly the provision services such as food, raw materials, and cultural service such as recreation. For unmarketable services, a price may be derived through indirect methods that take a surrogate market, or a hypothesized market as reference, such as travel cost, hedonic price, averting behavior, replacement/restoration cost, and contingent valuation methods (de Groot et al. 2002). Some ecosystem service can be valued by multiple methods and the selection of the appropriate method depends on the actual context. For example, water-supply services can be valued with the direct market, replacement cost, avoidance cost, travel cost, hedonic pricing, and contingent valuation methods. If the water supply is for drinking water, then the direct market method is the appropriate approach, but if water supply is for a waterfall or fountain, then the travel cost or hedonic method is the suitable method. If the water supply service is for controlling floods that are detrimental to the environment and human well-being, then the replacement/restoration method is applicable. It is important to disentangle the complex connections between different valuation methods. Take the travel cost and hedonic pricing methods as examples: the distance to the sea is closely correlated with air pollution, i.e., houses near the sea have lower levels of air pollution as well as lower travel costs. The hedonic method incorporates all of the influences of distance from the sea, including air pollution, scenic amenities, and access to the beach. In other words, the travel cost has been included in the hedonic valuation. A separate travel-cost study for access to the houses would possibly lead to double counting of the recreational part of the amenities with the hedonic effect analysis of distance.

Recommendations for reducing double counting

The complexity and underlying mechanisms of ecosystems make it difficult to completely remove double counting from ecosystem service valuations. We recommend disentangling the complex interrelations and establishing a concrete plan before ecosystem service valuation. The plan should address the spatio-temporal scale, ecological complexity, social capital traits (internal and external products), and the selection of appropriate valuation methods within the given context. Double counting occurs across different scales, in most cases, not due to the scale dependence of ecosystems per se, but to the fact that ecosystem *services* are scale dependent. The selection of the scale for valuation of ecosystem services is the first and most crucial step. The appropriate scale should be selected to reflect key feedbacks within ecosystems and correlations between ecosystems (Zhang et al. 2007). In particular, prior attention should be given to the causal relations between ecological mechanisms (processes), ecological phenomena (functions), and their direct and indirect contribution to human welfare (services). Human needs should be taken as the criterion of final benefits when selecting ecosystem services as a valuation index (Wallace 2007). Particular care is needed in relation to regulation and supporting services that are more susceptible to double counting than other services. Last but not the least, ecosystem service valuation methods are not independent of each other, and the selection of methods should consider the context of the research. If a particular individual valuation method is highly susceptible to double counting, then particular attention should be paid to it—the hedonic pricing and travel cost methods would be in this category (Fig. 2).

In order to better understand the suggestions of reducing double counting in ecosystem service valuation, we take the Yanhe Watershed of northern Shaanxi Province of China as an example. The valuation of ecosystem services in the Yanhe Watershed can be carried out in the following steps. First, the research scale should be specified in advance. With an area of 7,700 km^2, the Yanhe Watershed should be valuated at the landscape scale (Table 1). Second, the types of dominant ecosystem services at the landscape scale should be determined. The selection of ecosystem service types can be made on the basis of ecosystem mechanisms, human needs, and economics analysis. The major ecological problems facing the Yanhe Watershed are water loss, soil loss, biodiversity loss, and environmental degradation (Fu et al. 2005). Therefore, soil and water conservation, biodiversity conservation, and NPP (net primary productivity) can be selected as the most suitable ecosystem services. In this case, water regulation, nutrient retention, pollination, and air regulation should not be counted as they are intermediate processes and not directly sought by people. From the perspective of economics, the ecosystem services of soil and water conservation, biodiversity conservation, and NPP production are non-exclusive and have low overlap, so the selection of these ecosystem services has the lowest incidence of double counting. In relation to valuation methods, soil and water conservation can be valued by substituting valuation methods, e.g., building costs for reservoirs and biodiversity conservation can be valued by the contingent valuation method with carefully planned surveys of local residents' 'willingness to pay', while NPP production can be valued by direct market methods.

Conclusions

Ecosystem service valuation plays a significant role in environmental policy-making. The inherent characteristics of interdependence within ecosystem services mean that their valuation is more complex than simply valuing each category and summing them (Daily 1997). The characteristics of both externality and internality of ecosystem services means their valuation has many challenges. If the complex interrelations underlying double counting are not disentangled, ecosystem service valuation cannot provide accurate and logical information for policy-making. The causes of double counting for ecosystem services are not independent, and this makes the classification system of ecosystem services difficult. Ambiguity in the definitions of *ecosystem service* and *ecosystem function* results in a high likelihood of overlaps between different ecosystem services. The spatio-temporal scale dependency of ecosystem services brings the risk of double counting to the valuation process. The overlap between valuation methods and their inherent complex mechanisms also leads to double counting in the valuation process. The fundamental causes of double counting can be ascribed to the inherent properties of ecosystem services, complex ecosystem mechanisms, and scale dependency. Clarifying the differences between ecosystem functions, ecosystem processes, and ecosystem services helps to understand the mechanism and reduces the risk of double counting.

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