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ECOSYSTEM SERVICES, VALUES, AND SOCIETAL PERCEPTIONS OF INTERMITTENT RIVERS AND EPHEMERAL STREAMS

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ECOSYSTEM SERVICES, VALUES, AND SOCIETAL PERCEPTIONS OF INTERMITTENT RIVERS AND EPHEMERAL STREAMS

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- Intermittent rivers and ephemeral streams (IRES) contribute multiple ecosystem services (ES) that are not as widely recognized or appreciated compared with those of perennially flowing waters.
- Different ES are provided by IRES at different stages of their flow regime although the inherent uncertainty of, for example, flow may influence the way people value such services.
- Also undervalued is the spatial arrangement of ES provision by a river network that comprises perennial and IRES; the suite of ES in a region is probably more relevant than those provided by individual IRES alone.
- Economic valuation should consider the use and nonuse values of different IRES and during different flow phases.
- Strategies to enhance society's appreciation of the ES provided by IRES should translate into improved legislative protection, including restoration of IRES and their ES.

5.2.1 INTRODUCTION

Ecosystem services (ES) are the benefits that people obtain that are directly attributable to the ecological functioning of ecosystems (de Groot et al., 2002). Considering ecosystems from this perspective helps practitioners understand human relationships with nature, set management priorities, and formulate environmental policies (Carpenter et al., 2009; Daily et al., 2009; Seidl, 2014). There are many ways of classifying ES (Box 5.2.1), such as the classification by the Millennium Ecosystem Assessment (MEA, 2005a) that recognizes four broad types of ES: provisioning, regulating, supporting, and cultural services. Although this classification has several serious drawbacks (Box 5.2.1), it has been widely used for the last decade and distinguishes the direct and indirect links of ES that subsequently affect their valuation. For example, provisioning services have direct links to human needs for nutrition, shelter, or safety and are relatively easy to quantify economically (Costanza et al., 2014), whereas regulating and supporting services usually have more complex links with human needs and are less easy to quantify and value. Although most cultural services are readily linked to human values and are often used to raise public support for protecting ecosystems (Gobster et al., 2007; Daniel et al., 2012), many of these ES cannot be readily quantified in monetary terms. Nonetheless, there is growing demand for their explicit incorporation into ecosystem management and environmental policy agendas (Carpenter et al., 2009; Mace, 2014).

The current paradigm underpinning ecosystem management aims at ensuring sustainable provision of ES to society while maintaining the integrity, ecological function, and biodiversity of natural and, increasingly, novel ecosystems (Hobbs et al., 2014; Mace, 2014). This paradigm has evolved through several conceptual developments over the past two decades. After the importance of ES for sustaining human well-being was initially articulated and started to become popular (Daily, 1997), the next major conceptual development was the classification of ES into groups (e.g., MEA, 2005a; reviewed in Box 5.2.1). These classifications helped practitioners organize the diversity of different ES and communicate their relationships to resource managers, politicians, and the general public. This heralded the next conceptual development which entailed economic evaluations of each ES using various systematic approaches. One example is The Economics of Ecosystems and Biodiversity (TEEB, 2010) that extended the seminal work by Costanza et al. (1997) to value the economic significance of different ES by quantifying their monetary value in different ecosystems. The latest conceptual development has been the integration of ecological understanding and economic valuation of ES into a unified perspective that is rapidly gaining importance at local, regional, and global policy levels. Recent initiatives include the establishment of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) and the incorporation of ES in the 2020 targets set by the 10th Conference of Parties to the Convention on Biological Diversity (Carpenter et al., 2009; Larigauderie and Mooney, 2010; Matzdorf and Meyer, 2014).

In aquatic ecosystems, these conceptual developments of the ES paradigm have been evident in the increased focus on ES in strategies for management and conservation over the last two decades (Green et al. 2015; Boulton et al., 2016). All rivers and streams provide multiple ES, ranging from the supply of water for household, agricultural, and industrial uses through to the mitigation of flood and drought damage, and the provision of esthetic and recreational values (Brauman et al., 2007; Palmer et al., 2009). However, research on the provision and valuation of ES has focused almost solely on perennial rivers and streams; in stark contrast, ES and their values in intermittent rivers and ephemeral streams (hereafter, IRES) have been largely overlooked (Boulton, 2014). This oversight is surprising considering the prevalence of IRES on Earth (Chapter 1) and the physical, functional, and biological links between IRES and perennial waterways (Acuña et al., 2014; Dahm et al., 2015). Moreover, the global distribution of IRES is expanding owing to drying climates and increasing human demands for fresh water, and many once-perennial rivers are now intermittent (Larned et al., 2010; Gleick and Palaniappan, 2010; Jaeger et al., 2014). Such changes in flow regimes are likely to alter the provision of ES at local and landscape scales, especially when the biodiversity and ecosystem processes that underpin various ES are altered by increased frequency and duration of intermittence (e.g., Chapters 3.2, 4.3, and 4.10).

In the first part of this chapter, we explore the types of ES likely to be provided by IRES and how these ES might differ over time when conditions fluctuate between flowing and nonflowing phases, including loss of surface water. The second part of this chapter deals with different techniques to evaluate ES in IRES based on monetary and nonmonetary values. This valuation step is a crucial part of the ES paradigm (Haines-Young and Potschin, 2010). However, deciding which technique to use is challenging because all valuation techniques are social constructs and their "currency" is entirely dictated by human value systems and social attitudes (Larson et al., 2013; Sukhdev et al., 2014). Ignorance of the values of ES provided by IRES may be largely responsible for the widespread environmental degradation of these ecosystems (Boulton, 2014), and we urge systematic efforts to evaluate the ES of IRES globally. Our chapter concludes with speculation about the potential benefits of considering IRES within the framework of the current ES paradigm and how this perspective might guide wiser management and conservation of these undervalued ecosystems.

5.2.2 WHAT ECOSYSTEM SERVICES ARE PROVIDED BY IRES?

Although the concept of valuing natural ecosystems for their ES has been well established for over two decades (e.g., Costanza et al., 1997; Daily, 1997), there does not appear to have ever been a concerted effort to explicitly list and assess all of the ES of IRES as there has been for perennial streams and rivers. Indeed, despite considerable published research on ES, especially during the last decade, it seems that the application of the ES paradigm to IRES has been largely overlooked. For example, a literature search (Web of Science, October 15, 2015) on the topic words "ecosystem service* AND river*" yielded over 2000 titles whereas a search on “ecosystem service* AND intermittent river*” yielded only 16 titles. A closer look at these 16 publications indicates that most references to ES are tangential and none of these publications includes an attempt to list the ES of IRES or compare them with those in perennial rivers and streams.

In addition to societal ignorance leading to undervaluation of IRES (Section 5.2.4), there are other probable reasons why a list of ES explicitly for IRES has not yet been attempted. First, identifying and allocating ES is complicated by major inconsistencies in how different ES have been defined in the literature (see review in Nahlik et al., 2012). Although the four broad ES posited in the highly influential MEA (2005a) report are heuristically relevant, there is still no accepted approach for consistently identifying individual goods and services across studies of different ecosystems (Landers and Nahlik, 2013; La Notte et al., 2015). This lack of unification currently confounds the use of a universal and complete “checklist” of well-defined specific ES to apply to IRES. Instead, different countries have adopted different classifications (e.g., the UK National Ecosystem Assessment (UK NEA, 2011); the Final Ecosystem Goods and Services Classification System (Landers and Nahlik, 2013) in the United States; the Spanish National Ecosystem Assessment (SNEA, 2014)). Second, even where specific ES can be identified, establishing clear linkages between ecosystem functions and human well-being remains elusive (Ringold et al., 2013) because of our incomplete understanding of the mechanisms and processes that provide ES in many natural ecosystems. This is especially true in IRES where considerable uncertainty surrounds the influence of flow intermittence on many of the ecosystem processes that underpin ES (e.g., Chapters 3.2, 4.9, and 4.10). In addition, there is limited understanding of the interactions among these processes before and after flow ceases in IRES which further confounds assessment of the linkages between ecosystem functions and the provision of different ES at different stages of the flow regime.

Given these problems, we have adopted a very conservative approach to list the likely ES of IRES provided at different phases of their flow regime. This conservative approach acknowledges the many knowledge gaps about IRES ecosystem processes that currently hamper unequivocal allocation of ES and avoids the detailed subdivision of individual ES evident in other approaches (e.g., Landers and Nahlick, 2013). Therefore, despite its limitations (Box 5.2.1), we adopted the MEA’s (2005a) heuristic classification of provisioning, regulating, supporting, and cultural services to categorize the ES provided by or derived from “wetlands.” The MEA report defined wetlands in their broadest sense according to the Ramsar Convention on Wetlands which includes all perennial and nonperennial flowing and nonflowing waters. For the analyses in this chapter, we used the list tabulated in MEA (2005b) because this report focuses specifically on aquatic ES. The MEA (2005b) list omits some ES (e.g., transportation, use in mining) and suffers the broader constraints associated with the MEA (2005a) classification (Box 5.2.1). However, the list has the major advantages that (1) it covers the primary ES, (2) is a peer-reviewed and widely accepted table, and (3) was developed by an international team of respected scientists explicitly for application to wetlands defined in the broadest sense to include IRES.

To encompass the influence of flow intermittence and surface drying in IRES, we extended the MEA’s (2005b) table of ES to identify which services continue to be provided when flow ceases and when surface water disappears, including some hypotheses about how different ES are altered and whether additional ES might result (Table 5.2.1). Our table provides the basis for exploring the landscape-level provision of ES that result from the “spatial mosaic” of flowing, nonflowing (pool), and surface-dry channels typical of IRES at different stages of the flow regime and of linked perennial-intermittent river networks and their adjacent riparian zones, floodplains, and alluvial groundwaters.

Table 5.2.1 ES provided by or derived from IRES				
		Provision according to flow phase		
Ecosystem service	**Examples**	**Flowing**	**Pools**	**Dry**
Provisioning				
Fresh water	Surface water for domestic, industrial, and agricultural use	+	+ (but may be water quality issues)	Lost (or relies on access to subsurface water)
Food	Production of fish, wild game, fruits, and grains	+	+	Reduced or altered (some vegetation may derive water from groundwater)
Fiber and fuel	Production of logs, fuelwood, peat, and fodder	+	+	Reduced or altered (some vegetation may derive water from groundwater)
Biochemical	Extraction of medicines and other materials from biota	+	+ (if available in lentic biota)	Altered (may be derived from terrestrial biota)
Genetic materials	Genes for resistance to plant pathogens, ornamental species, etc.	+	+ (if available in lentic biota)	Lost (unless from biota that can use groundwater)
Regulating				
Climate regulation	Source of and sink for greenhouse gases; influence local and regional temperature, precipitation, and other climatic processes	+ (flow pulses may affect efflux of greenhouse gases)	+ (sediment OM important for sequestering carbon)	Altered (heat released or stored by dry channel may alter local air temperatures and humidity)
Water regulation (hydrological flows)	Groundwater recharge/discharge	+	Reduced (groundwater recharge may be reduced through loss of advection)	Lost (from surface sources)
Water purification and waste treatment	Retention, recovery, and removal of excess nutrients and other pollutants	+	Reduced or altered (loss of flow removes physical component of nutrient removal)	Reduced or altered (loss of water alters biogeochemical processes, likely reducing some purification processes)
Erosion regulation	Retention of soils and sediments	– (flow usually removes soils and sediments)	+	+ (dry channel may be sink for sediments carried by wind and other erosion)

Continued

Table 5.2.1 ES provided by or derived from IRES—cont'd

Ecosystem service	Examples	Provision according to flow phase		
		Flowing	Pools	Dry
Natural hazard regulation	Flood control, storm protection	+	+	+ (dry channel plays important role as sink for floodwaters; recharge for alluvial aquifers)
Supporting				
Soil formation	Sediment retention and accumulation of organic matter (OM)	− (flow usually removes sediments and OM)	+	+ (dry channel may be a sink for sediments and OM, especially as OM decomposition is slowed by drying (Chapter 3.2))
Nutrient cycling	Storage, recycling, processing, and acquisition of nutrients	+ (flow promotes nutrient spiraling; flow pulses affect cycling and storage (Chapter 3.2))	+	Reduced or altered (nutrient processing is slowed by drying; storage and microbial uptake altered in dry sediments)
Cultural				
Spiritual and inspirational	Many religions attach spiritual and religious values to aspects of wetlands; source of inspiration	+	+	Reduced or altered (values associated with water are lost but replaced by values associated with dry channels and gorges)
Recreational	Opportunities for recreational activities	+ (water-sports, fishing, etc.)	+ (water-sports, fishing, etc.)	Altered (walking, riding, etc., in dry channel)
Esthetic	Many people find beauty or esthetic value in aspects of wetland ecosystems (ecotourism)	+	+	Reduced (most people prefer channels containing water; however, dry gorges also attract tourists)
Educational	Opportunities for formal and informal education and training	+	+	+ (dry channels as "terra incognita")

The ES, arranged according to the four broad categories proposed by MEA (2005a), are ones listed by MEA (2005b) as provided by or derived from wetlands. For many of them, their provision (+ =provided; − =absent) varies with flow phase at the surface (defined as "flowing," "pools," and "dry"). Specific examples are discussed further in the text.

Assessments of ES must be considered from a broad perspective that explicitly acknowledges spatial variation in hydrological connectivity (Chapter 2.3) within individual IRES as well as across multiple river networks, both perennial and nonperennial.

Provisioning services from surface waters are typically lost or reduced by drying in IRES (Table 5.2.1) although alluvial water may be pumped from shallow aquifers below and along the channel. Obviously, where surface water is directly required for an ES, the service is lost when the IRES dries. However, because water quality often deteriorates when flow ceases (Chapter 3.1), some provisioning services are reduced during the nonflowing pool phase as well, even though surface water remains. It is likely that this collective loss or diminution of provisioning services underpins the societal undervaluation of IRES (Section 5.2.4) because the provisioning ES, particularly fresh water for domestic, agricultural, and industrial uses, are the most obvious ones to the public. This differential public valuation of provisioning services over regulating and supporting ES is also true for perennial streams and rivers (e.g., Gutiérrez and Alonso, 2013). Variations in hydrological phases not only alter the ecosystem processes that give rise to ES, but they can also change the accessibility to the goods derived from provisioning services. For example, in IRES, sediment extraction (Chapter 5.1) is easiest when flow has ceased and surface water is absent but flow is required to replenish mined stores via sedimentation and other geomorphological processes (Chapter 2.1).

Regulating services are lost, altered, or promoted by drying (Table 5.2.1). Recharge of shallow alluvial groundwater ceases when surface water disappears. However, the dry channel has an enhanced capacity to act as a sink for floodwaters (Fig. 5.2.1a and b) and sediments, helping to regulate the effects of erosion and natural hazards such as flooding. The pulsed flows characteristic of most IRES result in wide variation in some of the regulating ES provided during different flow phases. A promising area for future research on ES in IRES would be to assess how pulsed flow and intermittence affect the dynamics of greenhouse gases associated with aspects of local and global climate change. It is likely that the alternating redox conditions and other biogeochemical changes wrought by flow pulses (Chapter 3.1) mediate other crucial ES beyond climate regulation and the supporting service of nutrient cycling.

Provision of many supporting and cultural services also varies according to flow phase. Most are promoted during the flowing and pool phases but reduced or altered during dry phases (Table 5.2.1). For example, nutrient spiraling occurs during the flowing phase but when flow ceases, the downstream transport of nutrients in surface water ceases. When the channel dries, many of the physical and microbially mediated processes involved in nutrient cycling cease or become much slower (Amalfitano et al., 2008; Arce et al., 2014). Dry gorges of IRES are often major tourist attractions and the diverse other cultural services provided by dry channels are becoming better documented (Steward et al., 2012). When flowing, many IRES hold significant cultural values, especially in semiarid and arid regions (e.g., for indigenous peoples in central and northern Australia, Finn and Jackson, 2011).

There are two important points to consider when comparing the ES of IRES with those of perennial streams and rivers. The first is how the spatial arrangement of IRES and permanently flowing channels in a river network interact to affect the provision of ES by the different systems. The provision of many ES, especially those within the regulating and supporting categories (MEA, 2005a), is probably enhanced in both types of systems when intermittent and perennial sections intergrade. This enhancement arises from the diversity of different environmental conditions provided by the "spatial mosaic" at the landscape and catchment scale. For example, the ES of flood- and erosion-regulation (Table 5.2.1) are likely to be optimized where perennial sections can feed into IRES whose dry channels serve to buffer erosive effects and maximize bed surface area for recharge of shallow alluvial aquifers with excess floodwater (Chapter 2.3).

The spatial arrangement of linked intermittent and perennial stream reaches also favors the life cycles of some biota. Juvenile coho salmon (*Oncorhynchus kisutch*) with access to an intermittent tributary in an Oregon river grew faster in winter than fish restricted to the perennial channel, and this tributary also harbored some of the highest densities of spawning salmon in November-December (Wigington et al., 2006). The importance of the spatial arrangement of these linkages to the provision of different ES also extends to the adjacent riparian zone and underlying alluvial groundwater of IRES. Currently, we have scant knowledge of how ES provision is governed by the spatial arrangement, connectivity, and edge dynamics of "patches" of channels with flow durations ranging from perennial to ephemeral, and this would be another promising area for future research. There is increasing interest in developing catchment-scale restoration strategies in, for example, semiarid river basins (Trabucchi et al., 2014) that recognize the spatial arrangement of river network components when establishing approaches to enhance the delivery of ES at this scale and that prioritize patches (e.g., subcatchments) for restoration according to their potential to deliver the optimum combination of key ES for the entire basin.

This leads to the second important point to consider: ES do not operate in isolation but in suites of "bundles" of co-occurring processes. These bundles of ES result in collective outcomes, requiring the negotiation of trade-offs among the desired benefits for different stakeholders (Raudsepp-Hearne et al., 2010). When comparing the provision of ES between perennial and IRES, it is more logical to compare bundles of ES than to address them individually. This is especially true for management of sociological ecosystems because optimizing one ES without affecting the provision of others is unlikely and unrealistic (Seppelt et al., 2011; Berry et al., 2015). Palmer et al. (2014) provide a typical example of this trade-off of ES bundles that occurs during restoration strategies targeting incised channels of low-order perennial, intermittent or ephemeral stream reaches in urban areas. In an effort to maximize the bundle of ES including the reduction of bank erosion and promotion of the retention of nutrients and suspended sediments, the channel is converted into a stormwater management structure designed to reduce peak flows and enhance hydraulic retention. Although this design modifies the hydrological responses during some storm events and has potential to achieve the ES of sediment retention, there is no consistent pattern of nitrogen retention or removal that would lead to net annual benefits (Palmer et al., 2014).

Finally, flow regimes and ecological conditions have been so altered in various IRES in regions such as California and the Iberian Peninsula (Arthington et al., 2014) that they now support novel ecosystems. These novel ecosystems often harbor new combinations of species and, potentially, different ecological processes from those in natural IRES, with likely implications for the provision of ES. Where alterations to flow regime, water quality, and biota are so severe that restoration back to near-natural conditions is no longer feasible, there may be impairment or even loss of particular ES. This must be assessed when undertaking the sorts of "reconciliation ecology" advocated by Moyle (2014) for restoring ES in severely altered IRES. A promising future research direction is to ascertain how the proliferation of novel ecosystems will influence provision of different ES in river networks, including hybrid systems of naturally and artificially intermittent reaches. It is possible that these novel ecosystems may have intermittent flow yet not provide the same sorts of ES as natural IRES because of crucial differences in biodiversity or the ecosystem processes that underpin the provision of various ES. Furthermore, the ES of novel ecosystems may differ from natural ecosystems in their monetary and nonmonetary values.

5.2.3 VALUING THE ES OF IRES

Putting values on ES is a challenging and controversial exercise, and economists have often been criticized for attempting to put a price tag on nature (Silvertown, 2015). However, agencies responsible for managing or conserving natural resources must decide how to allocate scarce resources of money and time (Boulton et al., 2016), often involving trade-offs as discussed earlier. As these are economic decisions based either explicitly or implicitly on society's values, some form of economic valuation is needed to justify priorities and strategies for rational management and protection programs. Economic valuation provides monetary measures of the value of ES relative to other goods and services on which individuals spend their disposal income (Farber et al., 2006).

The economic value of benefits of ES can be classified into "use" and "nonuse" values (Pearce et al., 2006). Use values are those that are derived from the actual direct (e.g., water provisioning) and indirect use (e.g., fishing and bird-watching) of an ES. Nonuse values are those that do not actually

Table 5.2.2 Relationship between the MEA (2005a) categories of ES (Table 5.2.1) of IRES and their use and nonuse economic values

Category of ecosystem service	Types of values				
	Use values		Nonuse values		
	Direct	Indirect	Option	Existence	Bequest
Provisioning	√		√		√
Regulating		√	√		√
Supporting		√	√		√
Cultural	√		√	√	√

Cultural services encompass existence value, reflecting the benefit that humans enjoy from knowing that a resource exists. Box 5.2.2 describes the different types of economic values. Ticks indicate the type of values that may relate to the four ES.

consume an ES (e.g., esthetic value), and can be classified into option, existence, and bequest values (Table 5.2.2). Option value refers to the utility placed on maintaining or preserving a good, even if there is no likelihood of using it in the future (Pearce et al., 2006). Existence values relate to the benefit that individuals obtain from just knowing that an ecosystem and its ES exist. Finally, bequest values refer to the utility that individuals attach to ensuring that future generations will also enjoy benefits from ES.

Economic benefits from ES do not come without cost. Maintaining the level of provision of such services involves opportunity costs that concern the alternative uses of IRES. When policy options are being assessed, both costs and benefits should be considered in a cost-benefit analysis (Currie et al., 2009). Costs can be considered as “negative benefits.” From this perspective, lower levels of an ES are seen as yielding lower economic benefits or imposing financial costs to society for maintaining higher levels of that ES.

Table 5.2.2 shows how each ES relates to at least three different categories of economic values. For example, although direct use values are only associated with provisioning and cultural services of IRES, humans may place nonuse values (e.g., option and bequest values) on the same ES because these benefits are also obtained. However, supporting ES are only linked to nonuse values because humans do not actively consume such services. The same could be claimed for regulating services; however, as these include flood and climate change regulation, indirect use values are also potentially relevant. Option and bequest values can be attributed to all services provided by IRES because these ES represent potential demand for which full information might not currently be available. If these values were better known and publicized, individuals might be more inclined to protect IRES in light of the wide range of known and anticipated ES instead of causing irreversible changes through activities such as, for example, mining (Chapter 5.1) to exploit a short-term and transient ES, an activity that is likely to impair provision of many other ES in the future.

Economic value does not relate to the stock of ES that is available, but to increases or decreases in the level of provisioning of ES that cause changes in the magnitude of socioeconomic benefits. Fluctuation in the level of provided benefits will create variation in the value derived from ES. However, the total value of environmental goods cannot only be attributed to ES, but also to capital made by humans because, in some cases, environmental goods are the result of combining natural and manufactured capital such as infrastructure (Bateman et al., 2011). Following this argument, although we can assign economic values to various ES, it should not be regarded that if an ES ceases to exist, individual users’ utility decreases to zero (as might be the case when the flow of IRES decreases or stops). Such a belief could lead to overstating the values of ES and cast doubt on the accuracy of economic assessment. Therefore, before attempting to estimate the economic values of IRES, it is important to identify all relevant ES. This is because the type and magnitude of use and nonuse values differ among different IRES. For example, IRES far away from residential areas would be expected to generate lower direct use values compared to those closer to where individuals reside, due to higher costs of directly using the resources (e.g., water abstraction) when greater distances are involved for access to the ES. As IRES are often more abundant in less populated areas (e.g., arid and semiarid zones, Ortega et al., 2013), it is rational to assume that nonuse values are of high importance for such ecosystems. Although most individuals may live far away from IRES, they are still likely to attribute bequest and option values to them.

The identification of the relevant types of values helps to obtain robust results and defines the choice of the most relevant economic valuation technique (Table 5.2.3). This is because not all economic valuation techniques are able to elicit both use and nonuse values (Garrod and Willis 1999; Pearce et al., 2006). On one hand, *revealed preference techniques* (RP), a family of techniques that utilize the relationship between individuals' behavior and ES by using information taken from surrogate markets, can only estimate the use values of a resource (e.g., estimates of the recreational value of IRES based on the price of properties in the vicinity), but not the nonuse values attached to it. On the other hand, *stated preference techniques* (SP) are able to elicit both use and nonuse values, obtained by giving questionnaires to a representative sample of stakeholders. For this reason, SP techniques such as contingent valuation and choice modeling are more appropriate to estimate the total economic value of IRES than RP.

Table 5.2.3 The three main families of economic valuation techniques, their main application and notes on their relevance for valuing ES of IRES

Technique	Examples	Main application	Relevance for valuing ES of IRES
Revealed preference (RP) techniques (e.g., Bark-Hodgins, 2005)	Hedonic pricing Travel cost	Able to value changes in the status of IRES that have already taken place	Market data are required. These techniques are not able to estimate the value of supporting services (e.g., soil formation, nutrient cycling)
Stated preference (SP) techniques (e.g., Koundouri et al., 2014b)	Contingent valuation – Choice modeling – Choice experiment – Choice ranking	Able to estimate changes in the status of IRES that have taken or hypothetically could take place	These are more costly and time- and data-intensive measures but are able to monetize the benefits of all ES of IRES due to their survey format
Value transfer techniques (e.g., Koundouri et al., 2016)	Unit transfer Function transfer Meta-analysis	Can be used similarly to any of the previously mentioned. Initial studies might involve either RP or SP or both techniques	These techniques are easier to implement and are able to value all ES of IRES if these have previously been valued for other areas

Revealed preference techniques use market to value changes in IRES that have already occurred. On the other hand, stated preference techniques are based on data from surveys and can be used for ex ante (hypothetical changes) and ex post assessments. Value transfers can value both actual and hypothetical changes in IRES.

As IRES nonuse values are probably highly significant (discussed in detail later), successful planning of policies (Chapter 5.3) should be based on information about both use and nonuse values. One implication of failing to do so could be that use values would be favored over nonuse values, leading to inadequate economic valuation and, in worst-case scenarios, seriously compromising the future integrity of IRES and their capacity to provide other ES. Besides RP and SP techniques, *value transfer techniques* are also commonly used (Table 5.2.3). These belong to a different family of techniques that use results from earlier primary studies (either RP or SP) in other areas similar to the area under investigation. To account for socioeconomic differences, several adjustments (e.g., income differences, levels of prices, currency) are needed. Finally, other techniques exist, such as *averting behavior* based on the use of market prices to estimate the cost of investments to prevent an unwanted environmental change from occurring and *replacement cost* which is the estimated cost to replace an ES. Although several examples of the last two techniques are provided later, this chapter mainly focuses on the first three families of techniques: RP, SP, and value transfer techniques.

Aside from provisioning services, economic values associated with the other types of ES can be monetized using a number of techniques. Fig. 5.2.2 presents the associations between different types of values, ES and valuation techniques. Special attention is needed for valuing regulating and supporting

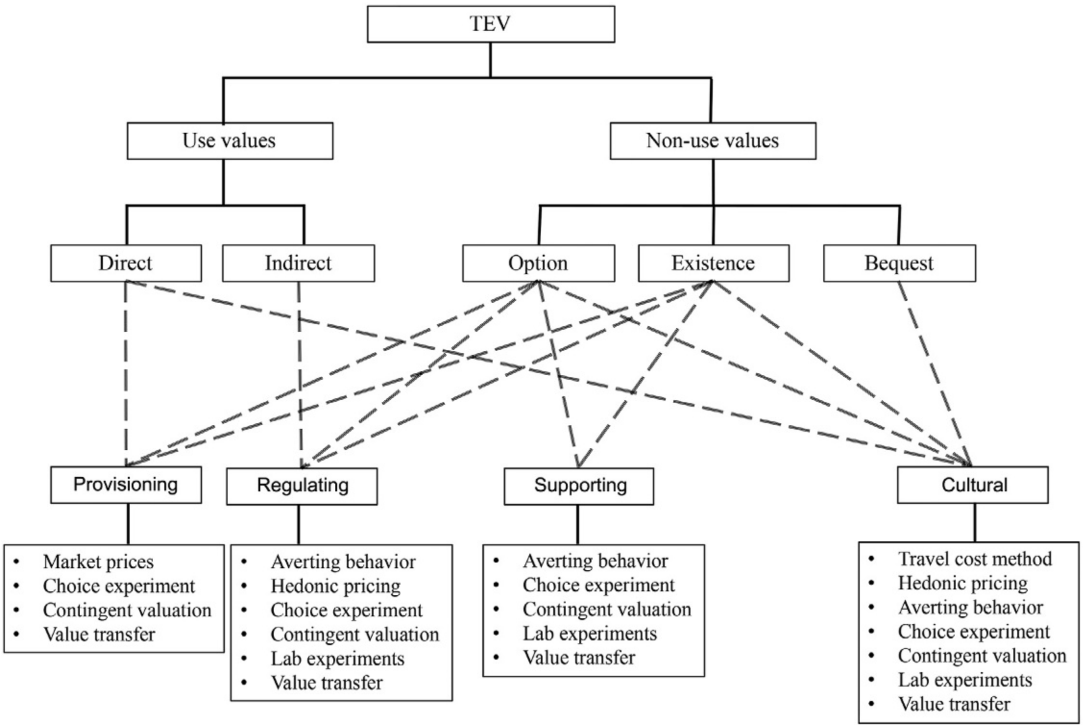

FIG. 5.2.2

Total economic value of ecosystems (TEV). The dashed lines denote the connections between the categories of economic value and the four main types of ES proposed by MEA (2005a). Under each ES category, a box presents examples of the many economic techniques that could be used to estimate economic value.

services. These types of ES are seldom adequately understood and, as a result, barely taken into account when policies are designed (Emerton et al., 2002). Generally, they are underestimated (Wood et al., 2010) and, as a consequence of this underestimation, other ES that depend on regulating and supporting processes might be threatened.

Given variations in the current management practices and the natural variability in morphological, chemical, and ecological status of IRES, economic valuation should be concerned with the impact of these changes on human welfare due to changes in provision of ES (Pagiola et al., 2004). When the ES of interest are related only to use values, RP techniques (such those mentioned earlier) are appropriate (Fig. 5.2.2). For example, Acuña et al. (2013) estimated the value of several ES in IRES using information from actual markets. These authors used the market price of brown trout (*Salmo trutta*) for estimating the value of the provision of fish and the mean market price of fishing permits was used to infer the value of opportunities of recreation. Additionally, the cost of replacing ES with technology (replacement cost technique) was used to estimate the benefits of water purification and the avoided cost technique (based on the investment cost to avoid a negative impact) was used to calculate the value of erosion control. Although several techniques were used, none of them was able to account for the nonuse values associated with four streams in the Añarbe reservoir in Spain. Further to the previously mentioned array of techniques, there are integrated models such the InVEST model (an open-source suite of tools used to map ES). These have been used to investigate the changes in the provision of ES and subsequent changes in value and land uses (e.g., Nelson et al., 2009; Sánchez-Canales et al., 2012; Bangash et al., 2013); however, in most applications, only use values are considered.

Hedonic pricing is an RP technique that has been used extensively. This technique enables the user to trace the footprint of the value of ES by observing a surrogate market. For instance, in two areas that have similar socioeconomic characteristics and differ only in the ES provided by the IRES, the value attributed by individuals to higher provision of ES can be assessed by comparing the prices of the properties in the two areas while keeping all other variables constant. Colby and Wishart (2002) used this technique to estimate the value of the Tanque Verde Wash in northeast Tuscany, which relates to ES associated with the scenic view, wildlife, and buffer from noise and pollution. House prices drop 0.45% for each 1% increase in distance from Tanque Verde, illustrating the spatial patterns of values of these ES. Another hedonic pricing study of ES in IRES (Bark-Hodgins, 2005) indicated that home-buyers assigned a higher value to the parts of the river where water flowed perennially or intermittently than ephemerally, due to the increased vegetation in the perennial and intermittent reaches. A final example is presented in Box 5.2.2 outlining the techniques used to estimate the economic value of several services provided by the Asopos river basin (Koundouri and Papandreou, 2014).

One key issue is the distinction between the total economic value of IRES that dry to pools and those that dry completely (broadly, the distinction between “intermittent” and “ephemeral”, Chapter 1). To the authors’ knowledge, there has not been a study that explicitly compares the economic values of the ES of these two types of IRES. However, the existence of pools secures higher levels of certain ES and their economic values than those present when the river is completely dry and some ES are even lost (Table 5.2.1). It is logical to assume that when accounting for nonuse values, the economic value of an IRES is higher during its pool phase than the dry phase, but further research is required on the economic importance of nonuse values during the dry periods. Transitioning between wet and dry phases may rejuvenate some ES that are less prevalent in perennial streams (e.g., some of the biogeochemical processes favored by wetting and drying, Chapter 3.2), and this has implications for the choice of valuation techniques that might correspond to the values associated with the different phases of IRES.

If valuation techniques focus on provisioning services (e.g., fresh water) that are lost when the river dries, then techniques heavily based on market data will give a different and potentially less accurate perspective than SP that account for both use and nonuse values (Table 5.2.3). Despite this, the accuracy of SP has often been criticized due to their susceptibility to biases.

Finally, when the goods of various ES of IRES are traded in a market and thus their price is indicative of their value, all valuation techniques can be used for the monetization of relevant benefits from provisioning, regulating, and most cultural services (Koundouri et al., 2016; Table 5.2.4). Exceptions may be the supporting services and some benefits accruing from cultural services. For the estimation of generated economic value, RP should be avoided since the market fails to reveal the demand for such services because they are provided for free by IRES and therefore people are unlikely to state how much they would be willing to pay for them. Consequently, the price that would allow the internalization of

Table 5.2.4 Four types of ES (from MEA, 2005), their goods and benefits, and the nonmarket techniques used to quantify their economic values		
Ecosystem services	**Benefits (goods and services)**	**Nonmarket techniques for the quantification of economic values**
Provisioning	These ES are related to goods that are of direct use for humans. Examples include water from an IRES that is used in agriculture as production input	Revealed preference Stated preference Value transfer
Regulating	These ES can be thought as benefits that control natural phenomena that help sustain or improve human life	Revealed preference Stated preference Value transfer
Supporting	These ES are vital for other ES, including ES in other ecosystems. For example, supporting ES in IRES can benefit riparian ecosystem services	Stated preference Value transfer
Cultural	These ES are nontangible goods that benefit human well-being	Revealed preference Stated preference Value transfer

Three families of nonmarket techniques (Table 5.2.3) can be used to estimate the benefits of ES in IRES.

externalities cannot be determined. To tackle this problem, SP and value transfer techniques should be used to avoid disregarding components of nonuse values. On the other hand, when the biggest share of the total economic value of the ES is attributed to use values, RP might be the best alternative. This argument is reinforced by the fact that the economic values of IRES are subject to great uncertainty arising from both uncertainty related to the timing and size of the flows as well as uncertainty and biases embedded in SP techniques. These associations between the four groups of ES proposed by MEA (2005a) and the families of economic valuation techniques are summarized in Table 5.2.4.

5.2.4 SOCIETAL PERCEPTIONS OF IRES

Societal decision-making and patterns of behavior are underpinned by perceptions (recognition and awareness of a state) and attitudes (evaluations of an object or outcome) (Kaiser et al., 1999). As IRES are readily identifiable parts of the landscape, it is reasonable to assume that most humans would recognize these ecosystems, even when dry, if the channels are well defined. When the waterways are flowing, people may not be aware that the waterways are IRES. However, as most IRES cease flow or are dry for over half the time, perceptions of IRES in the landscape can be assumed to be the norm for local landholders and most occasional visitors.

In contrast, attitudes toward IRES and their values are likely to be more diverse. Given the cryptic nature of many of the ES provided by these systems (Section 5.2.2), it would be predicted that most humans would undervalue IRES. This is especially true given the loss or reduction of provisioning services such as fresh water and food from surface waters when IRES dry (Table 5.2.1). In some areas, there is even an aversion to IRES because they are perceived as dangerous. For example, in many Mediterranean regions, the risk of flash-flooding within the channels and floodplains of ephemeral streams in peri-urban areas is mapped (e.g., Camarasa-Belmonte et al., 2011), and this is likely to influence land values and occupation patterns.

Consequently, legislation and incentives for management, protection, and restoration of IRES (Chapters 5.3–5.5) are strongly influenced by attitudes and, to a lesser degree, perceptions of these types of waterways. Surprisingly, few studies have been done to quantify human attitudes about IRES—a contrast to the rich literature on societal views on, for example, stream riparian buffer zones (e.g., Ryan et al., 2003; Kenwick et al., 2009). The most comprehensive study of human attitudes to IRES is by Armstrong et al. (2012) who interviewed landowners living along either perennial rivers or IRES in a small Pennsylvanian catchment about how "important" the stream was to them, how often it flowed ("always," "most of the time," "sometimes," "rarely"), together with other sociodemographic questions about age, education, length of ownership, and length of residence. Respondents were typically postmiddle-aged (mean = 62 years old) and had owned their riparian property for an average of 27 years. Those living along streams that flowed either always or most of the time rated the stream as significantly more important to them compared with the ratings of importance given by respondents living along IRES that flowed either "sometimes" or "rarely." Further data analysis indicated that the higher value given to perennial streams than IRES reflected a perception that water quality was better in the permanently flowing waterways (Armstrong et al., 2012), implying that land-owners' value judgments were based on provisioning ES for fresh water as well as perhaps cultural ES of recreation. The conclusions by Armstrong et al. (2012: p. 857) are blunt: "Landowner perceptions and attitudes reveal a disproportionate lack of concern towards ephemeral or intermittent streams."

Another line of evidence that IRES are less valued than perennial waterways is reflected by case studies of substantial demographic shifts when flow regimes have been changed. One example is a case study from the Água Limpa stream basin in the Jequitinhonha Valley of Minas Gerais, Brazil (Nogueira de Andrade and Leite, 2013). Over the last 50 years, stream flow (particularly base-flow during low rainfall periods) has declined associated with changes in land-use and management. Nogueira de Andrade and Leite (2013) claim that this increase in stream intermittence has been responsible for an exodus of rural residents to other parts of the country, resulting in marked socioeconomic changes in the region as rural percentages fell from 72% in the 1970s to only 53% in 1991.

One of the implications of societal attitudes and perceptions of IRES being lower than those of perennial waterways is the far greater possibility that channels of IRES will be used as convenient dumping grounds for rubbish (Fig. 5.2.3) and subjected to other intentional pollution and physical degradation (Chapter 5.1) less likely in the more-valued perennial waterways. A second implication is that the development of legislative protection for IRES will lag behind that for perennial waters (Acuña et al., 2014; Chapter 5.3). A third is the lower priority likely to be given to efforts to conserve or restore IRES, even by local landowners. Until there is a wider understanding of the ES provided by IRES leading to a change in public attitudes to these ecosystems, society will continue to undervalue IRES and our activities will continue to compromise many of the ES, especially the regulating and supporting ones listed in Table 5.2.1. Interestingly, these attitudes and perceptions of IRES closely resemble those reported from studies of perceived values of ES from terrestrial ecosystems in semiarid regions (e.g., Castro et al., 2014; Iniesta-Arandia et al., 2014). For example, in these terrestrial ecosystems, maintenance of water flow was considered to be the most important ES in both sociocultural and economic dimensions for all stakeholder

5.2.5 CONCLUSIONS AND PROGNOSIS

During the flowing phase, IRES likely provide very similar ES to those found in perennial rivers. Many of these provisioning, regulating, supporting, and cultural services persist when flow ceases and, in some cases, when surface water dries in the channel. The main difference between these ecosystems is that perennial rivers typically provide most of their ES at a more or less constant rate whereas in IRES, the marked changes in the flow regime and water permanence result in variable rates and provision of many ES. This variability and sometimes complete loss of ES, such as fresh surface water, as well as the current lack of understanding of supporting and regulating services leads to society valuing IRES less than nearby perennial rivers. Another reason why IRES are undervalued is that risk-averse individuals and management agencies perceive flash-flooding as being dangerous which impacts the land uses of areas within and near such ecosystems.

Economic techniques such as RP and SP are able to elicit societal preferences and attitudes towards IRES and to express the monetary value that individuals place on them. What is important is that different techniques should be used for the economic valuation of different ES of IRES. One of our main conclusions is that very few studies of ES and their values have been undertaken in IRES, corroborating the perception that IRES have been given little attention despite their ubiquity (Chapter 1). However, water scarcity caused by climate change and other stressors, such as overextraction of water (Chapter 5.1), will probably increase the prevalence and relevance of such ecosystems and their consideration by policy makers (Chapter 5.3). In particular, climate change gives rise to synergistic effects of multiple stressors, especially during periods of water shortage (Navarro-Ortega et al., 2015), an issue that is increasingly relevant in arid and semiarid areas. The significance of studying such poorly known ecosystems lies in the fact that the exploitation of use values risks jeopardizing or even losing nonuse values that could be relatively more important.

Faced with threats of water shortage through climate change and increasing human exploitation, coupled with the widespread misperceptions about IRES, one role of science should be to address such complex socioecological systems (Folke, 2006) and communicate results to relevant stakeholders who benefit from use and nonuse values of the diverse ES provided by intact IRES. Additionally, there should be more focus on the intangible benefits and ES of IRES, such as the existence value, because these values potentially comprise the biggest share of the total economic value of environmental resources (Johnston et al., 2003). Finally, scientific results should be integrated into policies in order to design measures that can efficiently raise awareness of the ES and economic values of IRES and, at the same time, create incentives for stakeholders to use these ecosystems in a more sustainable way. Educational programs on IRES and their importance for human welfare could help improve individuals' understanding and preferences in a way that protection of IRES could be better secured by targeted legislation and legal frameworks (Chapter 5.3).

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REFERENCES

Acuña, V., Díez, J.R., Flores, L., Meleason, M., Elosegi, A., 2013. Does it make economic sense to restore rivers for their ecosystem services? J. Appl. Ecol. 50, 988–997.

Acuña, V., Datry, T., Marshall, J., Barceló, D., Dahm, C.N., Ginebreda, A., et al., 2014. Why should we care about temporary waterways? Science 343, 1080–1082.

Amalfitano, S., Fazi, S., Zoppini, A., Caracciolo, A.B., Grenni, P., Puddu, A., 2008. Responses of benthic bacteria to experimental drying in sediments from Mediterranean temporary rivers. Microb. Ecol. 55, 270–279.

Arce, M.I., Sanchez-Montoya, M.D., Vidal-Abarca, M.R., Suarez, M.L., Gomez, R., 2014. Implications of flow intermittency on sediment nitrogen availability and processing rates in a Mediterranean headwater stream. Aquat. Sci. 76, 173–186.

Armstrong, A., Stedman, R.C., Bishop, J.A., Sullivan, P.J., 2012. What's a stream without water? Disproportionality in headwater regions impacting water quality. Environ. Manag. 50, 849–860.

Arthington, A.H., Bernado, J.M., Ilhéu, M., 2014. Temporary rivers: linking ecohydrology, ecological quality and reconciliation ecology. River Res. Appl. 30, 1209–1215.

Bangash, R.F., Passuello, A., Sanchez-Canales, M., Terrado, M., López, A., Elorza, J., et al., 2013. Ecosystem services in Mediterranean river basin: climate change impact on water provisioning and erosion control. Sci. Total Environ. 458, 246–255.

Bark-Hodgins, R.H., 2005. Do homebuyers care about the 'quality' of natural habitats? Doctoral dissertation, School of Life Sciences, Arizona State University, Phoenix, AZ.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner, K., 2011. Economic analysis for ecosystem service assessments. Environ. Resour. Econ. 48, 177–218.

Berry, P.M., Brown, S., Chen, M., Kontogianni, A., Rowlands, O., Simpson, G., et al., 2015. Cross-sectoral interactions of adaptation and mitigation measures. Clim. Chang. 128, 381–393.

Boulton, A.J., 2014. Conservation of ephemeral streams and their ecosystem services: what are we missing? Aquat. Conserv. Mar. Freshwat. Ecosyst. 24, 733–738.
Boulton, A.J., Ekebom, J., Gíslason, G.M., 2016. Integrating ecosystem services into conservation strategies for freshwater and marine habitats: a review. Aquat. Conserv. Mar. Freshwat. Ecosyst. 26, 963–985.
Brauman, K.A., Daily, G.C., Duarte, T.K., Mooney, H.A., 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. Annu. Rev. Environ. Resour. 32, 67–98.
Camarasa-Belmonte, A.M., López-García, M.J., Soriano-García, J., 2011. Mapping temporally-variable exposure to flooding in small Mediterranean basins using land-use indicators. Appl. Geogr. 31, 136–145.
Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., DeFries, R.S., Díaz, S., et al., 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc. Natl. Acad. Sci. 106, 1305–1312.
Castro, A.J., Verburg, P.H., Martín-López, B., Garcia-Llorente, M., Cabello, J., Vaugh, C.V., et al., 2014. Ecosystem service trade-offs from supply to social demand: a landscape-scale spatial analysis. Landsc. Urban Plan. 132, 102–110.
Colby, B.G., Wishart, S., 2002. Riparian Areas Generate Property Value Premium for Landowners. Agricultural and Resource Economics, College of Agriculture and Life Sciences, University of Arizona, Tucson, AZ.
Costanza, R., d'Arge, R., de Groot, R.S., Farber, S., Grasso, M., Hannon, B., et al., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.
Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., et al., 2014. Changes in the global value of ecosystem services. Glob. Environ. Chang. 26, 152–158.
Currie, B., Milton, S.J., Steenkamp, J.C., 2009. Cost–benefit analysis of alien vegetation clearing for water yield and tourism in a mountain catchment in the Western Cape of South Africa. Ecol. Econ. 68, 2574–2579.
Dahm, C.N., Candelaria-Ley, R.I., Reale, C.S., Reale, J.K., Van Horn, D.J., 2015. Extreme water quality degradation following a catastrophic forest fire. Freshw. Biol. 60, 2584–2599.
Daily, G.C., 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC.
Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., et al., 2009. Ecosystem services in decision making: time to deliver. Front. Ecol. Environ. 7, 21–28.
Daniel, T.C., Muhar, A., Arnberger, A., Aznar, O., Boyd, J.W., Chan, K.M., et al., 2012. Contributions of cultural services to the ecosystem services agenda. Proc. Natl. Acad. Sci. 109, 8812–8819.
De Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393–408.
Directive 2000/60. EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. Off. J. L 327, 1–73. http://eur-lex.europa.eu/legal-content/En/TXT/?uri=CELEX:32000L0060 Accessed 4 November 2016.
Drichoutis, A., Koundouri, P., Remoundou, K., 2014. A laboratory experiment for the estimation of health risks: policy recommendations. In: Koundouri, P., Papandreou, N.A. (Eds.), Water Resources Management Sustaining Socio-Economic Welfare. Springer, Netherlands, pp. 129–137.
Emerton, L., Seilava, R., Pearith, H., 2002. Bokor, Kirirom, Kep and Ream National Parks, Cambodia: Case studies of Economic and Development Linkages Field Study Report, Review of Protected Areas and their Role in the Socio-economic Development of the Four Countries of the Lower Mekong Region. International Centre for Environmental Management, Brisbane and IUCN - The World Conservation Union Regional Environmental Economics Programme, Karachi.
Farber, S., Costanza, R., Childers, D.L., Erickson, J., Gross, K., Grove, M., et al., 2006. Linking ecology and economics for ecosystem management. Bioscience 56, 121–133.
Finn, M., Jackson, S., 2011. Protecting indigenous values in water management: a challenge to conventional environmental flow assessments. Ecosystems 14, 1232–1248.
Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643–653.

Folke, C., 2006. Resilience: the emergence of a perspective for social–ecological systems analyses. Glob. Environ. Chang. 16, 253–267.

Garrod, G., Willis, K.G., 1999. Economic Valuation of the Environment: Methods and Case Studies. John Wiley and Sons, Chichester.

Gleick, P.H., Palaniappan, M., 2010. Peak water limits to freshwater withdrawal and use. Proc. Natl. Acad. Sci. 107, 11155–11162.

Gobster, P.H., Nassauer, J.I., Daniel, T.C., Fry, G., 2007. The shared landscape: what does aesthetics have to do with ecology? Landsc. Ecol. 22, 959–972.

Green, P.A., Vörösmarty, C.J., Harrison, I., Farrell, T., Sáenzc, L., Fekete, B.M., 2015. Freshwater ecosystem services supporting humans: pivoting from water crisis to water solutions. Glob. Environ. Chang. 34, 108–118.

Gutiérrez, M.R.V., Alonso, M.L.S., 2013. Which are, what is their status and what can we expect from ecosystem services provided by Spanish rivers and riparian areas? Biodivers. Conserv. 22, 2469–2503.

Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D.G., Frid, C.L.J. (Eds.), Ecosystem Ecology: A New Synthesis. Cambridge University Press, Cambridge, pp. 110–139.

Hobbs, R.J., Higgs, E., Hall, C.M., Bridgewater, P., Chapin, F.S., Ellis, E.C., et al., 2014. Managing the whole landscape: historical, hybrid, and novel ecosystems. Front. Ecol. Environ. 12, 557–564.

Iniesta-Arandia, I., García-Llorente, M., Aguilera, P.A., Montes, C., Martín-López, B., 2014. Socio-cultural valuation of ecosystem services: uncovering the links between values, drivers of change, and human well-being. Ecol. Econ. 108, 36–48.

Jaeger, K.L., Olden, J.D., Pelland, N.A., 2014. Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. Proc. Natl. Acad. Sci. 111, 13894–13899.

Johnston, R.J., Besedin, E.Y., Wardwell, R.F., 2003. Modeling relationships between use and nonuse values for surface water quality: a meta-analysis. Water Resour. Res. 39, 221–248.

Kaiser, F.G., Wolfing, S., Fuhrer, U., 1999. Environmental attitude and ecological behaviour. J. Environ. Psychol. 19, 1–19.

Kenwick, R.A., Shammin, R., Sullivan, W.C., 2009. Preferences for riparian buffers. Landsc. Urban Plan. 91, 88–96.

Koundouri, P., Papandreou, N.A., 2014. Water Resources Management Sustaining Socio-economic Welfare: The Implementation of the European Water Framework Directive in Asopos River Basin in Greece. vol. 7. Springer, Netherlands.

Koundouri, P., Papandreou, N., Stithou, M., Mousoulides, A., Anastasiou, Y., Mousoulidou, M., et al., 2014a. The economic characterization of Asopos River Basin. In: Koundouri, P., Papandreou, N.A. (Eds.), Water Resources Management Sustaining Socio-Economic Welfare. Springer, Netherlands, pp. 49–70.

Koundouri, P., Scarpa, R., Stithou, M., 2014b. A choice experiment for the estimation of the economic value of the river ecosystem: management policies for sustaining NATURA (2000) species and the coastal environment. In: Koundouri, P., Papandreou, N.A. (Eds.), Water Resources Management Sustaining Socio-Economic Welfare. Springer, Netherlands, pp. 101–112.

Koundouri, P., Rault, P.K., Pergamalis, V., Skianis, V., Souliotis, I., 2016. Development of an integrated methodology for the sustainable environmental and socio-economic management of river ecosystems. Sci. Total Environ. 540, 90–100.

La Notte, A., Liquete, C., Grizzetti, B., Maes, J., Egoh, B., Paracchini, M.L., 2015. An ecological-economic approach to the valuation of ecosystem services to support biodiversity policy. A case study for nitrogen retention by Mediterranean rivers and lakes. Ecol. Indic. 48, 292–302.

Lancaster, K.J., 1966. A new approach to consumer theory. J. Polit. Econ. 74, 132–157.

Landers, D.H., Nahlik, A.M., 2013. Final Ecosystem Goods and Services Classification System (FEGS-CS)EPA/600/R-13/ORD-004914. Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC.

Larigauderie, A., Mooney, H.A., 2010. The Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services: moving a step closer to an IPCC-like mechanism for biodiversity. Curr. Opin. Environ. Sustain. 2, 9–14.

Larned, S.T., Datry, T., Arscott, D.B., Tockner, K., 2010. Emerging concepts in temporary-river ecology. Freshw. Biol. 55, 717–738.

Larson, S., De Freitas, D.M., Hicks, C.C., 2013. Sense of place as a determinant of people's attitudes towards the environment: implications for natural resources management and planning in the Great Barrier Reef, Australia. J. Environ. Manag. 117, 226–234.

Mace, G.M., 2014. Whose conservation? Science 345, 1558–1560.

Matzdorf, B., Meyer, C., 2014. The relevance of the ecosystem services framework for developed countries' environmental policies: a comparative case study of the US and EU. Land Use Policy 38, 509–521.

Millennium Ecosystem Assessment, 2005a. Ecosystems and Human Well-being: Synthesis. World Resources Institute, Washington, DC.

Millennium Ecosystem Assessment, 2005b. Ecosystems and Human Well-being: Wetlands and Water Synthesis. World Resources Institute, Washington, DC.

Moyle, P.B., 2014. Novel aquatic ecosystems: the new reality for streams in California and other Mediterranean climate regions. River Res. Appl. 30, 1335–1344.

Nahlik, A.M., Kentula, M.E., Fennessy, M.S., Landers, D.H., 2012. Where is the consensus? A proposed foundation for moving ecosystem service concepts into practice. Ecol. Econ. 77, 27–35.

Navarro-Ortega, A., Acuña, V., Bellin, A., Burek, P., Cassiani, G., Choukr-Allah, R., et al., 2015. Managing the effects of multiple stressors on aquatic ecosystems under water scarcity. The GLOBAQUA project. Sci. Total Environ. 503, 3–9.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., et al., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and trade-offs at landscape scales. Front. Ecol. Environ. 7, 4–11.

Nogueira de Andrade, L., Leite, M.G.P., 2013. An analysis of the human activities impact on water quantity in the Jequitinhonha Valley, MG/Brazil. Manag. Environ. Qual. 24, 383–393.

Ortega, J.A., Razola, L., Garzón, G., 2013. Recent human impacts and change in dynamics and morphology of ephemeral rivers. Nat. Haz. Hydrol. Earth Syst. Sci. Discuss. 1, 917–956.

Pagiola, S., Von Ritter, K., Bishop, J., 2004. Assessing the economic value of ecosystem conservation. Environment Department Papers, Environmental Economics Number 101, World Bank, Washington, DC. https://openknowledge.worldbank.org/handle/10986/18391 Accessed 4 November 2016.

Palmer, M.A., Lettenmaier, D.P., Poff, N.L., Postel, S.L., Richter, B., Warner, R., 2009. Climate change and river ecosystems: protection and adaptation options. Environ. Manag. 44, 1053–1068.

Palmer, M.A., Filoso, S., Fanelli, R.M., 2014. From ecosystems to ecosystem services: stream restoration as ecological engineering. Ecol. Eng. 65, 62–70.

Pearce, D., Atkinson, G., Mourato, S., 2006. Cost-benefit analysis and the environment: recent developments. Organisation for Co-operation and Development (OECD), Paris.

Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. 107, 5242–5247.

Ringold, P.L., Boyd, J.W., Landers, D.H., Weber, M.A., 2013. What data should we collect? A framework for identifying indicators of ecosystem contributions to human well-being. Front. Ecol. Environ. 11, 98–105.

Ryan, R.L., Erikson, D.L., DeYoung, R., 2003. Farmers' motivations for adopting conservation practices along a riparian zone in a Midwestern agricultural watershed. J. Environ. Plan. Manag. 46, 19–37.

Sánchez-Canales, M., Benito, A.L., Passuello, A., Terrado, M., Ziv, G., Acuña, V., et al., 2012. Sensitivity analysis of ecosystem service valuation in a Mediterranean watershed. Sci. Total Environ. 440, 40–153.

Schwerdtner Máñez, K., Krause, G., Ring, I., Glaser, M., 2014. The Gordian knot of mangrove conservation: disentangling the role of scale, services and benefits. Glob. Environ. Chang. 28, 120–128.

Seidl, R., 2014. The shape of ecosystem management to come: anticipating risks and fostering resilience. Bioscience 64, 1159–1169.

Seppelt, R., Dormann, C.F., Eppink, F.V., Lautenbach, S., Schmidt, S., 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. J. Appl. Ecol. 48, 630–636.

Silvertown, J., 2015. Have ecosystem services been oversold? Trends Ecol. Evol. 30, 641–648.

SNEA (Spanish National Ecosystem Assessment), 2014. Ecosystems and biodiversity for human wellbeing. Synthesis of the key findings. Biodiversity Foundation of the Spanish Ministry of Agriculture, Food and Environment, Madrid.

Steward, A.L., von Schiller, D., Tockner, K., Marshall, J.C., Bunn, S.E., 2012. When the river runs dry: human and ecological values of dry rivers. Front. Ecol. Environ. 10, 202–209.

Sukhdev, P., Wittmer, H., Miller, D., 2014. The economics of ecosystems and biodiversity (TEEB): challenges and responses. In: Helm, D., Hepburn, C. (Eds.), Nature in the Balance: The Economics of Biodiversity. Oxford University Press, Oxford, pp. 135–152.

TEEB, 2010. The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations. In: Kumar, P. (Ed.), Earthscan, London, Washington.

Trabucchi, M., O'Farrell, P.J., Notivol, E., Comín, F.A., 2014. Mapping ecological processes and ecosystem services for prioritizing restoration efforts in a semi-arid Mediterranean river basin. Environ. Manag. 53, 1132–1145.

UK National Ecosystem Assessment (UK NEA), 2011. The UK national ecosystem assessment: synthesis of the key findings. http://uknea.unep-wcmc.org/ Accessed 4 November 2016.

Wigington, P.J., Ebersole, J.L., Colvin, M.E., Miller, B., Hansen, B., Lavigne, H., et al., 2006. Coho salmon dependence on intermittent streams. Front. Ecol. Environ. 4, 514–519.

Wood, M.D., Kumar, P., Negandhi, D., Verma, M., 2010. Guidance manual for the valuation of regulating services. UNEP, University of Liverpool, Liverpool. www.unep.org/pdf/Guidance_Manual_for_the_Regulating_Services.pdf Accessed 4 November 2016.

FURTHER READING

Apostolopoulos, K., 2010. Assessment of the Economic Importance of Water Use in the Industrial and Agricultural Areas of Asopos RB. Department of International and European Studies, Athens University of Economics and Business, Athens. http://www.aueb.gr/users/resees/en/econcharen.html (In Greek, Accessed 29.October 16).

Cummings, R.G., Taylor, L.O., 1999. Unbiased value estimates for environmental goods: a cheap talk design for the contingent valuation method. Am. Econ. Rev. 89, 649–665.

Fisher, B., Turner, R.K., 2008. Ecosystem services: classification for valuation. Biol. Conserv. 141, 1167–1169.

Kareiva, P., Watts, S., McDonald, R., Boucher, T., 2007. Domesticated nature: shaping landscapes and ecosystems for human welfare. Science 316, 1866–1869.

Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., Alkemade, R., 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biol. Conserv. 155, 1–12.

Ostrom, E., 2009. A general framework for analyzing sustainability. Science 325, 419.