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Analysis

An ecosystem services approach to estimating economic losses associated with drought

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ABSTRACT

A consistent methodology enabling the estimation of the economic losses associated with drought and the comparison of estimates between sites and across time has been elusive. In this paper, we develop an ecosystem service approach to fill this research gap. We apply this approach to analysis of the Millennium Drought in the South Australian portion of the Murray–Darling Basin which provided a natural experiment for the economic estimation of hydrological ecosystem service losses. Cataloguing estimates of expenditures incurred by Commonwealth and State governments, communities and individuals, we find that nearly $810 million was spent during the drought to mitigate losses, replace ecosystem services and adapt to new ecosystem equilibria. The approach developed here is transferable to other drought prone regions, providing insights into the potentially unexpected consequences of drought and ecosystem thresholds and socioeconomic and political tipping points after which ecosystem restoration may become very costly. Our application to the South Australian Murray–Darling Basin demonstrates the potential of this approach for informing water, drought preparedness and mitigation policy, and to contribute to more robust decision-making.

1. Introduction

The ecosystem services literature provides frameworks for cataloguing the multiple services provided by natural systems (Costanza et al., 1997; Millennium Ecosystem Assessment [MEA], 2003; The Economics of Ecosystems and Biodiversity [TEEB], 2010). These frameworks classify services as provisioning, regulating, habitat and cultural and amenity services. This paper is concerned with hydrological ecosystem services, the benefits that people derive from freshwater resources (Brauman et al., 2007), and how their supply was impacted by Australia's Millennium Drought (1997–2010).

Droughts affect more people than any other natural hazard (Wilhite, 2000) and can have severe and direct impacts on hydrological ecosystem services, for example affecting the supply and quality of water resources for municipal, industrial and agricultural use (Brown et al., 2002; Covich, 2009; Rosegrant, 1997). Droughts can also have important indirect effects, for example reduced water supply may force industrial users to reduce economic output thereby negatively affecting downstream industries (Mysiak and Markandya, 2009). Competing with these consumptive uses, riparian, wetland, and estuarine ecosystems require water to sustain them and the hydrological ecosystem services they provide.

Valuing natural capital in an ecosystem services framework can improve planning and decision-making (Daily et al., 2009) and enable more robust assessment of the ecological, socioeconomic and cultural trade-offs of ecosystem service loss (de Groot, 2010). Evaluating historical experiences of drought is one method to identify and estimate costs of hydrological ecosystem service losses. Greater understanding of drought impacts on ecosystem services can inform water reform, underpin efforts to support drought planning and adaptation (Covich, 2009), and provide a basis upon which budgets for mitigation and disaster assistance strategies may be prepared (Ding et al., 2010; Hayes et al., 2004; Mysiak and Markandya, 2009; Rose, 2004).

Evaluation of drought may reveal ecosystem sensitivities and thresholds: tipping points between two stable ecosystem states (Salt and Walker, 2006); and cascading or cumulative effects that may occur with ecosystem degradation (Kinzig et al., 2006). In addition to ecosystem thresholds, historical analysis of drought may reveal socioeconomic and political tipping points. A socioeconomic tipping point may be reached when, for example, agricultural losses result in crop insurance claims or prompt a community to lobby political representatives for support. A political tipping point may be associated with a biophysically-based trigger for policy response, for instance a water quality threshold. Better understanding of, and managing for, ecological thresholds and socioeconomic and political tipping points can provide water resource policy makers and managers with new information to avoid costly ecosystem degradation and tipping points.

Despite the importance of information on thresholds and tipping points, standard methodologies for estimating drought-induced hydrological ecosystem service losses are rare (Hayes et al., 2004). Considering the relatively lengthy duration of droughts, the slow pace at which they proceed and their spatial extent, quantification of associated economic

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impacts is complex (Ding et al., 2010). Furthermore, droughts differ from other natural hazards in that they often lack conspicuous disaster impacts (Hayes et al., 2004). However, in fully-allocated river basins such as the Murray–Darling, there can be observable drought-related ecosystem degradation (Crossman et al., 2011; Kingsford et al., 2011).

In this paper, we develop an approach to cataloguing and estimating the hydrological ecosystem services losses associated with drought. We demonstrate this approach using an ecosystem service framework combined with evaluation of thresholds and tipping points in the South Australian Murray–Darling Basin (SA-MDB). The case study provides specific estimates of ecosystem service losses, ecological thresholds and socioeconomic and political tipping points, which can inform water, drought preparedness and mitigation policy in the SA-MDB. In particular we focus on retrospective assessment of ecosystem service losses that are revealed through directly observable defensive, mitigation, rehabilitation expenditures and damage costs. The approach can be applied more generally where there is information on these costs.

In the case study we find that the approach developed provides a useful organizing principle for cataloguing and quantifying the economic impacts of hydrologic ecosystem service losses associated with drought as well as the identification of ecological thresholds and socioeconomic and political tipping points that trigger a societal response. The estimates generated here should be interpreted carefully, however, since they are not measures of economic welfare rather they are an estimate of the costs of drought-induced hydrological ecosystem service losses. Still, we find that studying the magnitude and timing of such costs retrospectively has the advantage of revealing place-specific thresholds and tipping points that may not otherwise be evident.

Following the introduction, a brief description of the study area is provided. Next, the methods section presents the conceptual underpinnings of ecosystem service analysis, the typology used to classify drought impacted ecosystem services, and the approach to estimating the costs incurred as a result of ecosystem service losses. The fourth section provides estimates of expenditure by ecosystem service type. In the discussion section we highlight the strengths and limitations of the approach. The paper concludes with some lessons learned in the application of the approach to the case study of the SA-MDB.

2. Study Area

Our study is focussed on the South Australian portion of the Murray–Darling Basin (Fig. 1), the outflow sub-catchment of the Murray–Darling Basin (MDB). The MDB occupies one seventh of the Australian continent and is multi-jurisdictional with portions of the watershed in the States of South Australia, Victoria, New South Wales, and Queensland, as well as the Australian Capital Territory. The country's longest rivers are located in the basin, namely the Darling, the Murray and the Murrumbidgee Rivers, and 2.1 million people (10% of the Australian population) reside in the MDB.

Both dry land and irrigated agriculture are economically important activities, accounting for 10% of employment in the MDB. The MDB contains 65% of Australia's irrigated agricultural land (ABS, 2008). Perennial and annual horticulture and rice are important irrigated crops in the southern basin (MDBA, 2010a). Irrigated agriculture's share of total consumptive water use in the MDB was 83% in 2004–05. The MDB also has important social (Bark, 2011; Bryan et al., 2011), cultural and indigenous (Jackson et al., 2010; Weir, 2009) and environmental values (Hatton MacDonald et al., 2011; Tapsuwan et al., 2012).

Since the 1930s, the volume of water extracted from the basin increased from 3000 gigalitres (GL) to 11000 GL in the 1990s (MDBA, 2010a). This growth in consumptive demand coincided with a time of high average inflows. The most recent drought, the Millennium Drought, was the most severe drought in recorded history (Potter et al., 2010), with water availability in the southern basin reduced to less than 40% of the long-term average by 2006–07 (MDBA, 2010a). As the drought deepened, storages were drawn down, irrigation water allocations declined to record lows (Kirby, 2011) and States temporarily suspended their water sharing plans (NWC, 2009).

Long-term over-allocation of water resources combined with inadequate adaptive drought management practices resulted in cascading spatial and temporal freshwater ecosystem degradation (Chiew et al., 2010; CISRO, 2012; Crossman et al., 2011; Kingsford et al., 2011). Examples of freshwater-dependent ecosystem degradation were standwide death of river red gums (MDBC, 2003), the formation of acid sulphate soils (Baldwin, 2011a), subsidence and river bank slumpage, hypersalinity in the lagoon complex situated at the terminus of the River Murray (Kingsford et al., 2011; Lester and Fairweather, 2011; Overton et al., 2010), and the siltation of the mouth of the River Murray (Webster, 2010).

Responding to the crisis, the Commonwealth and South Australian governments took defensive and mitigation measures as the Millennium Drought progressively deepened. These agencies commissioned studies on damage costs and preventative measures, and recorded expenditures. The declaration of riverbank collapse as a State hazard by the SA State Emergency Management Committee and the subsequent institution of South Australia's Riverbank Collapse Hazard Program is one such example.

3. Methods

The methods developed here involve three components. The conceptual framework underpinning the approach is the concept of ecosystem services cascades. The second component is the ecosystem service typology which enables the classification of ecosystem services. Third is the approach to estimating the costs incurred as a result of ecosystem service losses.

The concept of ecosystem service cascades (Fig. 2) was developed in Haines-Young and Potschin (2010) and modified in TEEB (2010). The core notion is that ecosystems and biodiversity are linked to human well-being and a cascade links the system components of the ecosystem service production chain. In this paradigm, structures and processes give rise to specific ecosystem functions which are physical, chemical or biological in nature (de Groot, 2010), and contribute to human well-being (Haines-Young and Potschin, 2010).

For example, a riparian ecosystem is an ecological structure. A key function of this system is the provision of shade to maintain a stable water temperature while one of the services this function provides is habitat for fish. The fish produced in this ecosystem provide recreation and food benefits to anglers. Governance arrangements influence how, when, and which fish may be harvested. These arrangements impact the management of the ecological system and therefore its functions. The ecosystem services framework relies on ecological, socio-cultural or economic metrics by which the impacts of ecosystem services on well-being are gauged. A distinction is made between the benefits of an ecosystem service and any monetary value placed on it. Ecosystem service benefits can be estimated by various environmental valuation techniques or may not involve the calculation of monetary value (Haines-Young and Potschin, 2009).

To enable classification and systematic analysis of ecosystem services, de Groot et al (2010) developed a typology which specifies the relationship between ecosystem structures, processes, functions, services and well-being.[1] This typology was later adapted by TEEB (2010) and is the classification system applied in this paper (Table 1). The first column of Table 1 presents the ecosystem processes which underpin the ecosystem services listed in column two. There are four main categories of ecosystem services: provisioning, regulating, habitat, and cultural and amenity services. Earlier typologies including that of the Millennium Ecosystem Assessment (2003) included supporting services, which

[1] See TEEB (2010) for a discussion of early developments in ecosystem services frameworks.