Ecosystem Services 21 (2016) 153–165

Contents lists available at ScienceDirect

Ecosystem Services

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

Is groundwater recharge always serving us well? Water supply provisioning, crop production, and flood attenuation in conflict in Wisconsin, USA

Eric G. Booth [a,*], Samuel C. Zipper [b], Steven P. Loheide II [c], Christopher J. Kucharik [d]

[a] Department of Agronomy, Department of Civil & Environmental Engineering, University of Wisconsin-Madison, USA
[b] Freshwater & Marine Sciences Program, Department of Civil & Environmental Engineering, University of Wisconsin-Madison, USA
[c] Department of Civil & Environmental Engineering, University of Wisconsin-Madison, USA
[d] Department of Agronomy, Nelson Institute Center for Sustainability and the Global Environment, University of Wisconsin-Madison, USA

ARTICLE INFO

Article history:
Received 14 February 2016
Received in revised form
31 July 2016
Accepted 9 August 2016

Keywords:
Hydrologic services
Groundwater
Recharge
Water supply
Crop production
Flooding

ABSTRACT

Ecosystem service mapping can provide an avenue for making effective land management decisions in a holistic way. However, mapped quantities do not always appropriately represent the ecosystem services that are used by humans. We highlight this issue with a case study of groundwater recharge, water supply, flooding, and agricultural production in an urbanizing agricultural watershed in southern Wisconsin, USA. Groundwater recharge is typically treated as a beneficial ecosystem service or service indicator whose value to humans monotonically increases with the amount of recharge. While appropriate from a water supply perspective, this relationship breaks down when excess groundwater recharge leads to flooding and crop damage. We suggest moving beyond groundwater recharge as a stand-alone ecosystem service, and instead propose that observations and biophysical models should be used to quantify the final service humans receive from groundwater (e.g. reliability of water supply from a municipal well). Integration of such derived, point-based metrics with other ecosystem services that are more easily represented at the landscape scale remains a challenge for regional ecosystem service inventories and analyses.

1. Introduction

Ecosystem service mapping is a rapidly growing field within ecology and sustainability science (Burkhard et al., 2012; Egoh et al., 2008; Kareiva et al., 2011; Qiu and Turner, 2013). It has been proposed as an integrated framework for robust science-based assessments of current and future landscapes to aid decision makers with respect to land-use planning, conservation, and climate change adaptation (Burkhard et al., 2013). Mapping of ecosystem services allows researchers and stakeholders to visualize spatial heterogeneity in supply and demand that can lead to geographically-specific management solutions (Crossman et al., 2013). However, many challenges remain regarding the best way to put the framework into practice due to inconsistent methods of mapping and accounting for multiple services that yields a composite assessment of a landscape (Polasky et al., 2015; Schaefer et al., 2015). A trend towards monetization of ecosystem services suggests that appropriate accounting methods that combine multiple services together on equal footing are critical, but current methods vary and are likely to be place-specific (Olander et al., 2015).

Many studies have investigated complex relationships among ecosystem services in an attempt to gain a more holistic understanding of a given system (Bennett et al., 2009; Qiu and Turner, 2013; Raudsepp-Hearne et al., 2010). Bennett et al. (2009) proposed two categories to organize these relationships: 1) common drivers such as land use change affecting multiple services and 2) interactions among services (e.g. synergies, tradeoffs). Here, we further explore interactions among services but specifically consider the case in which an ecosystem service indicator affects multiple services. We present groundwater recharge (a common ecosystem service indicator for water supply provisioning and water regulation) and its effects on agricultural production and flood attenuation as a special example of the first category in which a single driver, in this case a physical process, affects multiple services.

Groundwater recharge is a part of the ecosystem service group termed hydrologic services, which are commonly found in ecosystem service mapping and accounting assessments (Brauman et al., 2007; Guswa et al., 2014). While groundwater recharge is almost always presented in the context of water supply

* Corresponding author.

http://dx.doi.org/10.1016/j.ecoser.2016.08.007
2212-0416/

provisioning, our objective is to demonstrate its positive and negative relationship with agricultural production and flood attenuation/moderation. We use the Yahara River watershed in south-central Wisconsin as a case study not only to elucidate these complex connections but also discuss wider significance to the science of ecosystem services, particularly in humid areas throughout the world.

1.1. Differing concepts of ecosystem services

Ecosystem service mapping represents a specific application of the ecosystem services concept, which has advanced the ability of land management decision makers to account for the benefits that ecosystems provide for humans in a more holistic way. The popularity of this approach has dramatically increased in recent decades in the environmental science, management, and conservation fields (Dempsey and Robertson, 2012) and ecosystem service markets are being created with the intention of addressing pervasive issues including biodiversity loss, carbon mitigation, and water quality degradation (Kinzig et al., 2011; Kronenberg and Hubacek, 2013). During this rapid growth, the meaning and definition of 'ecosystem service' has been debated by many economists and environmental scientists (Boyd and Banzhaf, 2007; Fisher et al., 2009; Nahlik et al., 2012; Wallace, 2007; Wong et al., 2015). A major point of dispute is whether or not to include indirect contributions of ecosystems to human well-being, although other important debates exist about accounting for ecosystem 'disservices' (Shackleton et al., 2016), excluding services predominantly produced by human activities (Ringold et al., 2013), implicitly reinforcing certain economic norms (Ernstson and Sorlin, 2013), and not adequately accounting for social power relations and inequality (Berbés-Blázquez et al., 2016). The lack of consistent definitions and approaches has made regional-scale ecosystem service assessments that are relevant to decision makers very difficult (Nahlik et al., 2012; Turner and Daily, 2008). This is especially true when aggregating multiple ecosystem services that can interact in complex and non-linear ways (Bennett et al., 2009; Robertson et al., 2014).

One emerging ecosystem services framework is that which distinguishes between 'intermediate' and 'final' ecosystem services (Boyd and Banzhaf, 2007; Fisher et al., 2009; Johnston and Russell, 2011). Final ecosystem services are "components of nature, directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf, 2007, 619) while intermediate ecosystem services are "conditions or processes that only benefit humans through effects on other, final services" (Johnston and Russell, 2011, 2244). The translation of intermediate to final services requires the use of a biophysical (or ecological) production function model that utilizes intermediate services and other human and environmental factors as inputs and simulates the final services (Boyd and Banzhaf, 2007). This framework makes the ecosystem services concept operational in cost-benefit analyses and ecosystem service valuation activities by avoiding double-counting but is also being used in non-valuation ecosystem service aggregation approaches (Ringold et al., 2013). Supporters of this framework argue that a formal accounting structure that includes only final ecosystem services is necessary when accumulating benefits to assess cumulative changes in ecosystems and human well-being under various scenarios (Ringold et al., 2011; Wong et al., 2015).

This framework differs from some prominent studies that more broadly define ecosystem services to include both intermediate and final ecosystem services (Costanza et al., 1997; Daily, 1997; MEA, 2005). One reason for a more inclusive definition is the inability for biophysical production function models to accurately estimate final services given an incomplete scientific understanding of a process or cost of acquiring the necessary inputs and parameters (Costanza, 2008). In addition, they argue that avoiding intermediate services eliminates an opportunity to connect underlying drivers with the production of ecosystem services (Costanza, 2008). However, those advocating for only using final services in accounting and aggregation do state the importance of understanding intermediate services given they provide inputs to ecological production function models and represent opportunities for management interventions (Boyd and Banzhaf, 2007; Ringold et al., 2011).

1.2. Ecosystem service mapping

The specific application of mapping ecosystem services provides a good example of how these two differing viewpoints can impact a regional analysis of ecosystem services. A primary goal of ecosystem service mapping is to help decision makers evaluate tradeoffs between multiple ecosystem services and identify areas to concentrate management efforts to increase the total ecosystem service value produced within a region of interest (Kareiva et al., 2011). Most studies that attempt to map a suite of ecosystem services are forced to only quantify *indicators* of the services due to the lack of data or knowledge of the actual service, but these indicators based on readily available data are often inadequate for capturing the full concept of a particular ecosystem service (Mace and Baillie, 2007; Reyers et al., 2013). In this context, indicators are often synonymous with intermediate ecosystem services. The implicit assumption is that the indicator has a positive, monotonic relationship with the ecosystem service it intends to represent (Fig. 1A).

Furthermore, in mapping applications, the particular indicators are often chosen because they can be readily mapped across an area. Often, mapping some intermediate ecosystem services makes intuitive sense because it represents the variables that management can directly influence. However, that activity differs from the goal of a full assessment of ecosystem services, which requires an accounting of only the final services. Recent studies have recognized the challenge of mapping both intermediate and final services (e.g., Castro et al., 2015) but issues remain for how to properly integrate them.

1.3. Hydrologic services

Considerable attention in ecosystem service assessment and mapping efforts has been given to ecosystem services that are related to freshwater (Brauman, 2015; Garrick et al., 2009) because water is essential to human well-being and a healthy economy (Vigerstol and Aukema, 2011), and freshwater ecosystems are recognized as among the most vulnerable ecosystems in the world (Postel et al., 1996). Brauman et al. (2007) provide a review of these freshwater or hydrologic services and offer a classification system that includes the following categories: diverted water supply, in situ water supply, water damage mitigation, spiritual and aesthetic, and supporting. While each of these categories excluding supporting can potentially be considered final services, the operational definition and use of the term hydrologic services is often expanded to include items that are better defined as indicators of final services or intermediate services. These typically include the "supporting services" of water retention, water yield, and water filtration (Vigerstol and Aukema, 2011). However, if we apply the concept of final services to freshwater, the final hydrologic services of interest are quantities such as clean water provision and flood damage mitigation.

While human well-being is a complex and evolving concept, there is a general consensus that ecosystem services are necessary but not sufficient for human well-being (Butler and Oluoch-Kosura, 2006). Implicit with this argument is that the more

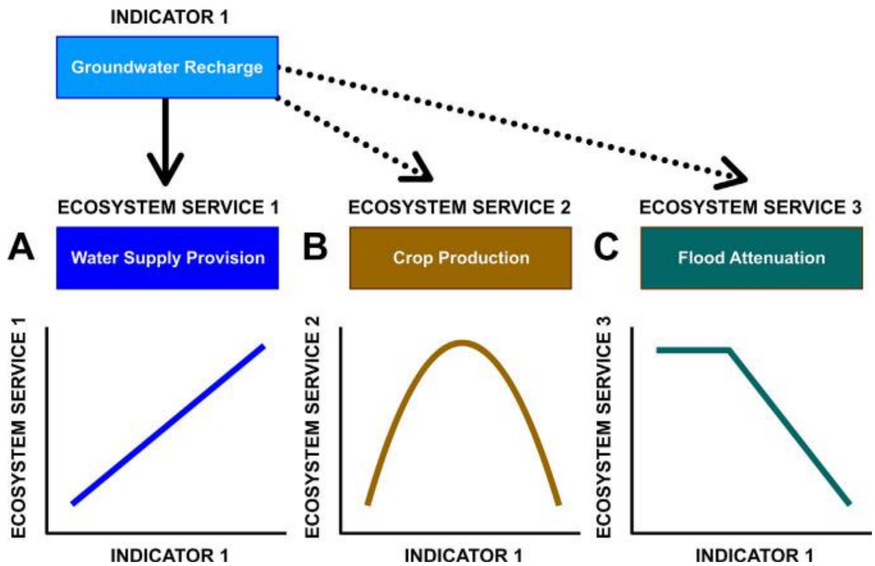

Fig. 1. Conceptual model of implicit assumption of positive, linear relationship between an ecosystem service (e.g. water supply provision) and a chosen indicator (e.g. groundwater recharge) to be mapped (A). However, the indicator also has a non-linear relationship with a second ecosystem service (crop production) (B) and negative relationship with a third (flood attenuation) (C). Thus, using groundwater recharge as an ecosystem service indicator can lead to confusion.

ecosystem services (or indicators of ecosystem services) that are acquired or made accessible, the more likely it is that human well-being will improve. This is clearly not the case with water. Hydrologic services are atypical of most ecosystem services because maximum human well-being is likely to occur at intermediate levels of hydrologic quantities. For instance, a river provides a service for navigation if the level is above a critical threshold that allows passage and below a higher threshold where passage becomes unsafe. Thus, we have historically attempted to engineer and moderate river flows to achieve this intermediate optimum. While others have suggested the relationship between hydrologic fluxes and ecosystem services can be non-linear (Guswa et al., 2014), the non-monotonic nature of this relationship has received little attention even though it has important implications for land management decisions.

1.4. Groundwater recharge as an ecosystem service

Groundwater recharge – water entering the saturated zone at the water table – is commonly considered an ecosystem service (Baral et al., 2013; Barbier, 2007; Burkhard et al., 2012; Euliss et al., 2008; Serna-Chavez et al., 2014) and it can be estimated and mapped using readily available climate and soil parameters (e.g. Dripps and Bradbury, 2007). However, groundwater recharge is only the entry-point to a complex groundwater flow system that includes both deep aquifers that municipalities tap into for water supply and unconfined aquifers with a shallow water table that can reach the ground surface and interact with plant root zones and influence runoff generation (Fig. 1).

From an ecosystem service mapping perspective, groundwater recharge is almost universally treated as a beneficial service because it increases groundwater supply for human consumption, maintains baseflow, and supports groundwater-dependent ecosystems (Fig. 1A). When considering water supply provisioning as the ecosystem service of interest, we find that implicit in this conceptualization is the direct link between recharge and an increase in aquifer storage at the point where water is pumped for human consumption. But more specifically, recharge leads to an increase in storage of the uppermost aquifer at the recharge location that can be quite removed in distance and travel time from pumping locations. This lag time represents an issue when recharge can also lead to detrimental impacts such as groundwater flooding and subsequent crop losses (oxygen stress in vegetation) and property damage (Nosetto et al., 2015; Zipper et al., 2015). While other ecosystem services that are commonly mapped are produced and consumed in situ (e.g. forest recreation and aesthetics), groundwater recharge belongs to a category where effects can be manifested much farther away in both space and time and can be both beneficial and detrimental to several final ecosystem services (as we document below).

While perceiving groundwater recharge as a beneficial service that provides adequate water supply to human and ecosystem needs is intuitive in regions where water supplies have been or are threatened to become diminished, this viewpoint does not necessarily hold in other regions where groundwater supplies are abundant and/or have been increasing due to increased precipitation (e.g., Motew and Kucharik, 2013) or land use change (e.g., Potter, 1991). A manifestation of excess groundwater supply is groundwater flooding, which occurs when the water table rises above the ground surface. Contrary to most surface-water flooding events, groundwater flooding can last for an extended period of time (weeks to years) and associated damages can be substantial (Kreibich and Thieken, 2008). Regions including the United Kingdom (Holman et al., 2009; Hughes et al., 2011), Germany (Kreibich and Thieken, 2008), the Argentina Pampas (Kuppel et al., 2015; Nosetto et al., 2009), Washington, USA (Jones et al., 2000) and the Upper Midwest, USA (Gotkowitz et al., 2014) have experienced substantial groundwater flooding in recent decades and research is ongoing related to climate change impacts on recharge in these regions (e.g. Holman, 2006). In addition, elevated groundwater levels following recharge events can make a landscape more vulnerable to river flooding by increasing the runoff response (Fig. 1C; Dunne and Black, 1970). Nosetto et al. (2015) document the increased landscape-scale flooding risk associated with more groundwater recharge under annual crops as opposed to perennial crops in the Pampas region of Argentina.

In urban areas, groundwater recharge is commonly viewed as a beneficial ecosystem service as it relates to offsetting the impacts of historical groundwater pumping and reduced recharge due to increased impervious area (Ferguson, 1990; Potter, 2006). These impacts include increased pumping costs and reduced flows to wetlands, streams, springs, and other groundwater-dependent ecosystems (Barlow and Leake, 2012; Hamel et al., 2013). This perspective has even been codified into government rules and regulations to encourage stormwater infiltration and recharge through best-management practices (Roy et al., 2008). However, groundwater recharge can also be linked to decreased depth to water tables and subsequent basement flooding and foundation damage in both urban and rural areas (Cadavid and Ando, 2013; Soren, 1976).

In some rural agricultural areas the view of groundwater recharge can also be mixed. With its clear connection to human well-being through provision of food, fiber, fuel, and pharmaceuticals, agricultural production is commonly considered an ecosystem service as production landscapes are increasingly recognized as agricultural ecosystems (Costanza et al., 1997; MEA, 2005; Power, 2010). Recent research has highlighted the importance of considering tradeoffs between agricultural production and other ecosystem services in managed landscapes (Baral et al., 2013; Bennett et al., 2009; Burkhard et al., 2012; Qiu and Turner, 2013). In agricultural areas, groundwater represents an important water source for irrigation, and enhanced groundwater recharge would help counteract groundwater abstraction in many overexploited regions globally (Gleeson et al., 2012; Wada et al., 2010). Furthermore, recharge can lead to shallow groundwater that can be beneficial in creating what has been described as a *groundwater subsidy* (Lowry and Loheide, 2010) for both crops and natural ecosystems when adequate moisture is not available from rain-fed infiltration or irrigation. Conversely, too much recharge can lead to elevated water tables that can be detrimental by creating waterlogged or oxygen-stress conditions in the root zone of crops leading to declines in productivity (Nosetto et al., 2015; Soylu et al., 2014). This decline in agricultural production leads to the concept that recharge can also be an ecosystem disservice that undermines or harms human wellbeing (Shackleton et al., 2016) as well as a service, and has been termed a *groundwater yield penalty* (Fig. 1B; Zipper et al., 2015). The non-monotonic relationship between the value to human well-being and recharge makes it inaccurate – even implicitly – to consider groundwater recharge and other hydrologic fluxes as universally beneficial services to humans.

1.5. Objective of case study

In contrast to the widespread assumption of a monotonic increase in a final ecosystem service with an increase in its indicator (e.g., Fig. 1A), we hypothesize that final ecosystem services will exhibit a variety of both positive and negative responses to increasing groundwater recharge (Fig. 1A–C). The objective of this paper is to show how groundwater recharge can lead to a service and disservice in the same watershed depending on climate, location, and ultimate beneficiary. Using the Yahara River watershed – an urbanizing agricultural basin in south-central Wisconsin – as a case study, multiple hydrologic datasets are synthesized to explain the influence of climate and urbanization on water resources and agriculture. This positions our study as an example of place-based research that is critical for advancing scientific research on ecosystem services by grounding new concepts and theories – such as the ones we present here – in real-world observations (Carpenter et al., 2009). Several recent studies have highlighted the importance of considering humans as integral components of socio-ecological systems (e.g., Vogel et al., 2015) and acknowledged that synthesis of varied data sources is necessary to untangle complex interactions between people and the ecosystems in which they reside (Liu et al., 2007). In this manner, we make a unique contribution to the ecosystem services literature by bringing together diverse ecological, hydrological, and economic lines of evidence to introduce and support a new conceptual model for understanding the services and disservices resulting from groundwater recharge.

2. Case study – Yahara River watershed

2.1. Study area

The Yahara River watershed in south-central Wisconsin is an urbanizing, agricultural watershed (Fig. 2A). The watershed lies mostly in Dane County but extends to Columbia and Rock counties to the north and south, respectively. Rain-fed corn, soybean, and alfalfa crops dominate the land cover in the northern and southern thirds of the basin and support a large dairy industry. The urban footprint in the center third has greatly expanded over the last 150 years, especially in the last several decades (Gillon et al., 2015; Wegener, 2001). Madison – with a metropolitan area population of 568,593 in 2010 – is the seat of government for the state of Wisconsin and is home to the University of Wisconsin–Madison.

Regional climate is characterized as sub-humid continental with mean annual precipitation of 837 mm. Previous studies have estimated regional recharge rates ranging from 64 to 287 mm with a mean of 230 mm (Hart et al., 2012). Watershed topography is heavily influenced by the Wisconsin Glaciation that resulted in a poorly drained landscape with four large lakes. In addition, internally-drained basins are common (Fig. 2B) and particularly susceptible to subtle changes in the hydrologic cycle. The groundwater flow system (Fig. 3) consists of essentially two aquifers – an early Cambrian sandstone (deep aquifer) and a Paleozoic bedrock/Pleistocene glacial deposit (shallow aquifer) – separated by a relatively continuous but leaky shale aquitard (Bradbury et al., 1999).

2.2. Methods

We integrate and synthesize disparate data sources including hydrological observations, local policy documents, and crop insurance data to reveal differing perceptions of groundwater recharge in the Yahara River watershed. To quantify changes in groundwater pumping and wastewater effluent discharges, we analyzed water use data compiled by the U.S. Geological Survey (*C. Buchwald*, unpublished) and wastewater effluent volume data from the Madison Metropolitan Sewerage District (*D. Taylor*, unpublished). Groundwater and streamflow data were downloaded from the USGS National Water Information System website (http://waterdata.usgs.gov/nwis) and used to analyze trends in water levels in the lower and upper aquifers, low flows in the Yahara River at McFarland, Wisconsin (Site no. 5429500), and peak annual streamflows in the Yahara River at Windsor, Wisconsin (Site no. 5427718). Data from NASA's Landsat 5 project was downloaded (http://glovis.usgs.gov/) and post-processed to determine inundation area for each image from 1983 to 2011 in the Upper Yahara River watershed. More details on the method for calculating inundation area are available in Appendix A.

We used simple linear regression and the *F*-test to determine if a trend is significant. We also used the two-sample Kolmogorov–Smirnov (K–S) test to determine if the sample probability distribution has changed from an earlier to a later period. This second test has been used to look at the effects of dam regulation on streamflow (Ren and Kingsford, 2014) and can determine if extreme events (floods, droughts) have changed with time.

Document analysis was performed to assess current stormwater management ordinances related to groundwater recharge implemented by various governance entities in the study area. Crop insurance indemnity and liability data for Dane County, Wisconsin (http://www.rma.usda.gov/data/cause.html) were also

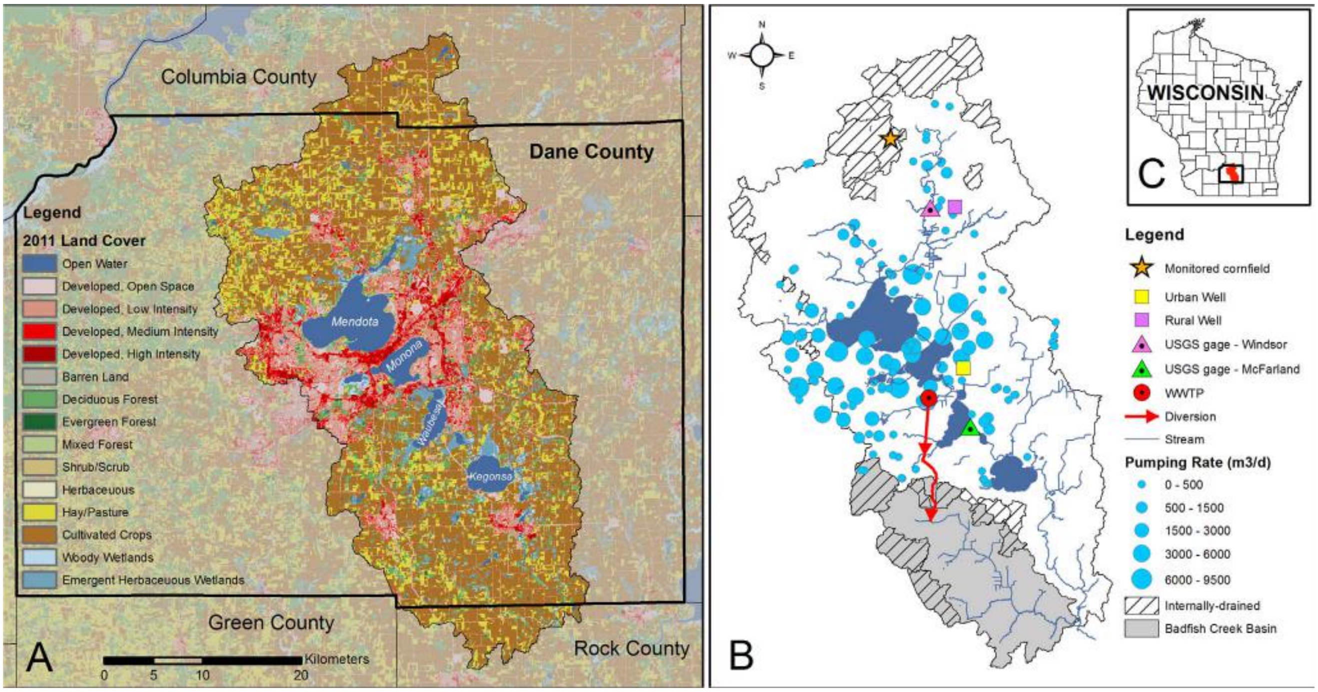

Fig. 2. Map of the Yahara River watershed land cover based on 2011 USGS National Land Cover Dataset (A). Municipal water transfer from groundwater pumping wells (blue circles) to the Nine Springs Wastewater Treatment Plant (WWTP) and diverted to the Badfish Creek Basin (gray); also showing locations of groundwater monitoring wells in urban (yellow) and rural (purple) areas, streamflow monitoring gauge (green and pink triangles), and internally-drained basins (hatched) (B). Location of the Yahara River watershed in state of Wisconsin, USA (C). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

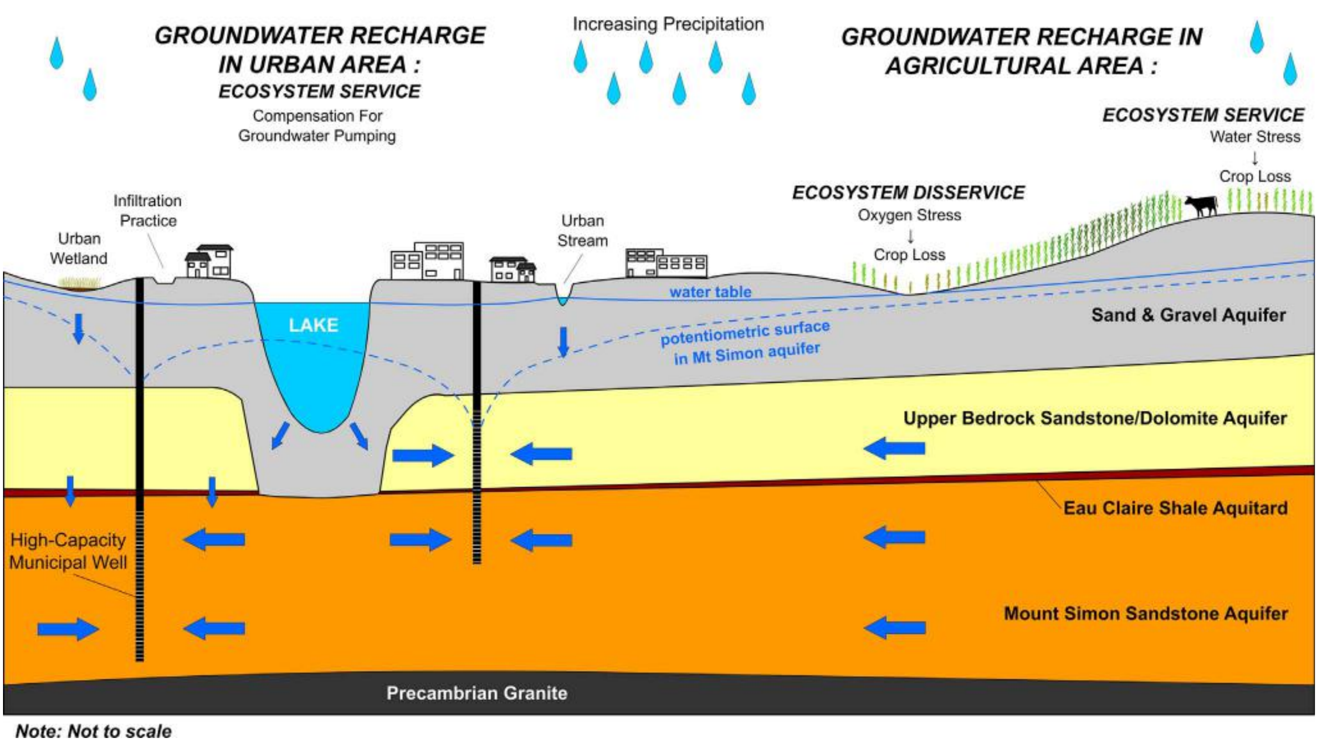

Fig. 3. Conceptual diagram showing the groundwater flow system in the Yahara River watershed and the different interpretations of groundwater recharge in urban and agricultural areas.

analyzed to show changes in payouts for causes of losses related to excess moisture and drought (USDA, 2016). The average ratio of indemnity to liability for two specific causes of loss (excess moisture and drought) were calculated for four periods of equal length from 1948 to 2015 (17-year) to assess change through time for these two crop-impacting hydrologic extremes. Biophysical data from a commercial corn field were also analyzed and presented to show how shallow groundwater can be both beneficial and detrimental in the same field. During the 2012 and 2013 growing seasons groundwater levels, leaf area index [LAI], and crop yield were monitored at two neighboring corn fields in the northern Yahara Watershed (Fig. 2B). Thermal imagery was also collected over each field in order to estimate spatial variability in evapotranspiration.

3. Results

3.1. Groundwater recharge in urban areas

As the population of the Madison metropolitan area expanded throughout the 19th and 20th centuries, groundwater withdrawals also increased for public water supply and industrial applications (Fig. 4). The vast majority of pumped water is removed from the high-yielding deep aquifer. Beginning in 1928, the system's wastewater was collected and treated at the Nine Springs wastewater treatment plant (WWTP) just south of Lake Monona. Treated effluent was originally discharged to the Yahara River upstream of Lake Waubesa but water quality concerns in the lower lakes eventually led to diversion of the effluent out of the mainstem Yahara River basin to the Badfish Creek basin in 1958 (Fig. 2B). Effluent discharge to Badfish Creek has tracked closely with municipal groundwater withdrawals in the urban area through time following diversion (Fig. 4).

This pumping and diversion system has led to lowering of water levels in the deep aquifer as observed and estimated since the 1970s (Bradbury et al., 1999; McLeod, 1978) (Fig. 5). Groundwater flow model simulations estimate maximum drawdowns in the deep aquifer of more than 60 feet in areas close to pumping with less near the lakes (Krohelski et al., 2000). This reduction in water levels has also reduced flows to springs, wetlands, and streams upstream of the diversion (Bradbury et al., 1999). Evidence of declines in flows to springs has been reported previously (Bradbury et al., 1999; Macholl, 2007) but data is limited. Historical spring survey data collected by the U.S. Geological Survey in 1958 and 1967 at Merrill Spring in the city of Madison shows a mean flow of 7.5 L s^{-1}. However, following construction of a municipal well within 0.5 km, 6 out of 8 measurements from 1975 to 1977 revealed zero flow (USGS, unpublished data). In addition, field and modeling studies have documented the sensitivity of spring flow to changes in pumping and recharge in the Yahara Basin (Hunt et al., 2001; Swanson et al., 2009). Concerns related to pumping impacts on wetlands have been present in the community since the 1960s (Baumann, 1968).

Previous research quantified the impact of groundwater pumping and diversion on streamflow in the Yahara River (Fetter, 1977; Young, 1966). The continued impact is best observed at the McFarland streamflow gage on the Yahara River (Fig. 2B) where annual 7-day minimum flows commonly fall below 0.4 $m^3\,s^{-1}$ following diversion (Fig. 6). The average annual frequency of these

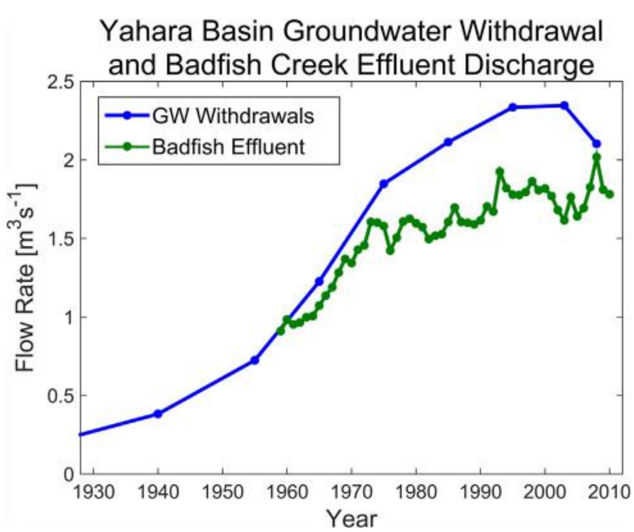

Fig. 4. Increasing trend in groundwater (GW) withdrawals in the Yahara River watershed and treated wastewater effluent discharge to Badfish Creek.

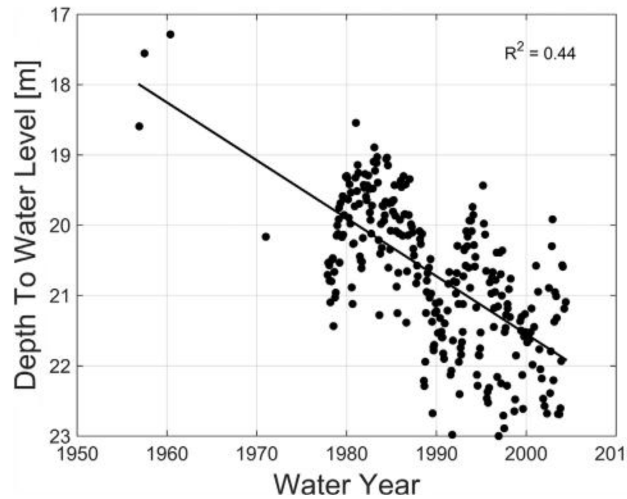

Fig. 5. Decreasing trend in the water level in a groundwater monitoring well screened in lower sandstone aquifer in urban area of Yahara River watershed. Trend is significant ($p < 0.001$) using F-test and Mann–Kendall test.

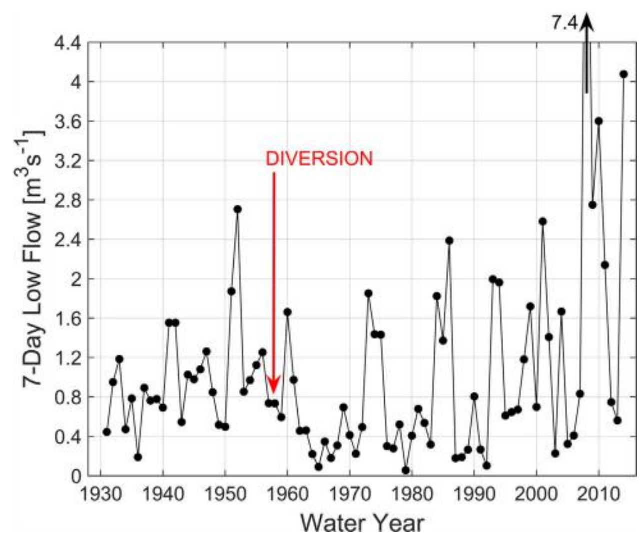

Fig. 6. Annual 7-day low flow in the Yahara River at McFarland, WI, showing an increase in the frequency of values below 0.4 $m^3\,s^{-1}$ following diversion of treated wastewater effluent to Badfish Creek in 1958. These lower values indicate the diversion of groundwater through the municipal system and eventually downstream of the gage. Higher values towards the end of the record indicate increased precipitation and groundwater recharge in the watershed.

very low flows ($< 0.4\ m^3\,s^{-1}$) prior to diversion was 0.04 and after diversion it rose to 0.30. In addition, the two-sample K–S test rejects the null hypothesis that pre-diversion data is from the same distribution as post-diversion data ($p < 0.05$).

While groundwater pumping was still substantial before diversion, the impact on streamflow was minimized during low flows by supplementing natural river discharge with treated effluent. However, after diversion the reach of the Yahara River downstream of the Madison metropolitan area and upstream of the confluence with Badfish Creek is now subject to the full impacts of pumping and reduced recharge from increasing impervious area. In response to low-flow concerns in the 1990s, regional water managers contracted with the U.S. Geological Survey to develop a plan to mitigate the impacts of low flow reductions by strategically releasing water from the regulated outlets of upstream lakes (Krug, 1999).

Efforts to address the impacts of pumping, diversion, and increased impervious surfaces have existed in the region for several decades and are best documented through changes in stormwater runoff and infiltration ordinances for new developments. While the majority of state, county, and municipal stormwater ordinances emphasize the goal of reducing stormwater runoff and its water quality impacts, concerns over reduced recharge and the overall change to the urban water balance (Ferguson, 1990) have led to more specific language related to infiltration and recharge. The state of Wisconsin, beginning in 2013, requires infiltration best-management practices (BMPs) to be implemented post-construction and includes provisions such as infiltration of at least 90% of the pre-development infiltration volume for an average year. This follows from the finding that "uncontrolled post-construction runoff…can degrade physical stream habitat by…diminishing groundwater recharge" (Ch. NR152B.S.02, Wis. Adm. Code). The stormwater ordinance for Dane County in 2013 provides another compliance requirement for new developments by meeting or exceeding the annual pre-development recharge rate as specified by estimates from the Wisconsin Geological & Natural History Survey (Ch. 14.51(2)(e)3, Dane Co. Ordinances). They also explicitly include "promote infiltration and groundwater recharge" as an ordinance objective (Ch. 14.43(2)(c), Dane Co. Ordinances). Municipalities within the county are allowed to further increase the required infiltration and/or recharge from new developments. For example, the Town of Westport within Dane County requires infiltration of 100% of the pre-development infiltration volume for an average year. In the context of these ordinances, it is clear that groundwater recharge in urban areas of the Yahara River watershed is viewed as a beneficial process and ecosystem service – even though it is not explicitly labeled as such – that reduces the impacts of pumping, diversion, and impervious area.

Even though the view of groundwater recharge in urban areas of the Yahara River watershed as a beneficial hydrologic flux that offsets the impact of development is implicit in many local ordinances and regulations, excess recharge can also lead to urban flooding issues that are separate from flooding concerns caused by increased impervious surfaces. Local newspaper reports have highlighted this groundwater flooding issue in recent years (Livick, 2013; Rickert, 2008; Simms and Leaf, 2007) but no total damage estimates have been compiled. Regardless, to those that are affected by basement flooding, groundwater recharge could conceivably be viewed as a detrimental process even though links to water supply may also be understood (Cadavid and Ando, 2013).

3.2. Groundwater recharge in agricultural areas

The view of groundwater recharge in regards to agriculture in the Yahara River watershed is quite different than that in urban areas. This is largely attributed to recent groundwater flooding caused by increasing precipitation and recharge. Although the frequency of very low flows in the Yahara River increased following wastewater effluent diversion in 1958, low flows have been increasing overall since 1990 due to an increasing precipitation trend in the watershed (Fig. 6). At the Madison airport weather station, annual precipitation has been increasing at an average rate of 2.5 mm/year from 1930 to 2014 (Fig. 7). This trend is consistent with others throughout the Upper Midwest (Baker et al., 2012; Pryor et al., 2009; Qian et al., 2007) and Wisconsin (Kucharik et al., 2010). Previous research has indicated increasing groundwater recharge driven by this positive precipitation trend in the Upper Midwest (Motew and Kucharik, 2013). It has also been shown that the region surrounding the Yahara River watershed has experienced higher than average recharge rates in 2006, 2007 and 2008 (Hart et al., 2012).

In response to increasing precipitation and groundwater recharge, groundwater levels in the upper aquifer have been increasing as observed in a well near the town of Windsor north of

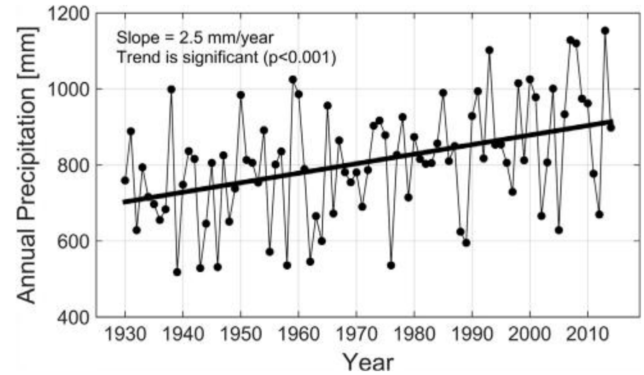

Fig. 7. Increasing trend in annual precipitation at the Madison airport.

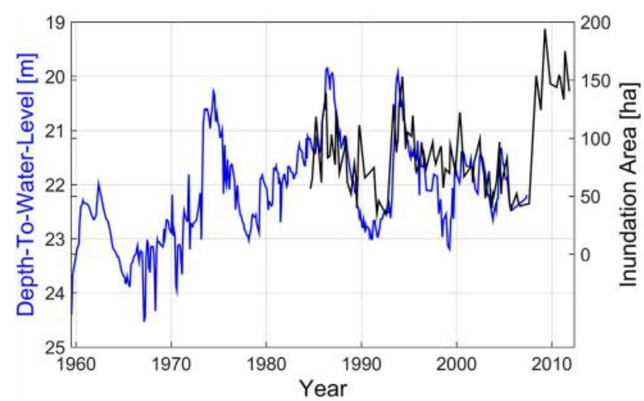

Fig. 8. Changes in the water level in the upper aquifer and inundation area (derived from Landsat imagery) in rural areas of the northern Yahara River watershed.

the main groundwater withdrawal zone around Madison (Figs. 2B and 8). The water table has increased above the ground surface (i.e. groundwater flooding) over a greater areal extent during this trend period as observed with Landsat imagery (Fig. 8). The groundwater level and Landsat-derived inundation area time-series are well correlated (Pearson $r = 0.60$). The increasing trend in groundwater level from 1959 to 2007 is statistically significant ($p < 0.01$) while the shorter increasing trend in inundation area from 1984 to 2011 is moderately statistically significant ($p < 0.1$). Crop yield and groundwater level data from a commercial corn field in the watershed suggests that the increasing area of inundation is negatively impacting agricultural operations in low-lying fields where oxygen stress leads to reductions in crop yield (Zipper et al., 2015). Increased area of inundation also reduces total planted area and excess moisture can delay spring planting, shortening the growing season and reducing yield. As evidence, $18,495,000 (in 2012 dollars) in federal crop insurance indemnities have been paid to Dane County farmers with "excess moisture" as the cause of loss from 1990 to 2012 (USDA 2016). The fraction of crop insurance liability that has been paid out as an indemnity to farmers as a consequence of excess moisture has increased from an annual average of 0.6% in the period between 1948 and 1981 to 1.4% from 1982 to 2015 (Fig. 9). While other social and institutional factors may play a role in this increase, the relatively large indemnity payments indicate a substantial cost to society.

Explicitly connecting the "excess moisture" cause of loss with groundwater flooding is challenging due to the data collection techniques of the USDA-RMA. However, it is a reasonable explanation because "flooding" is a separate cause of loss defined to represent surface water flooding. In the context of groundwater flooding and oxygen stress, groundwater recharge is linked with the concept of a *groundwater yield penalty* in areas with a shallow

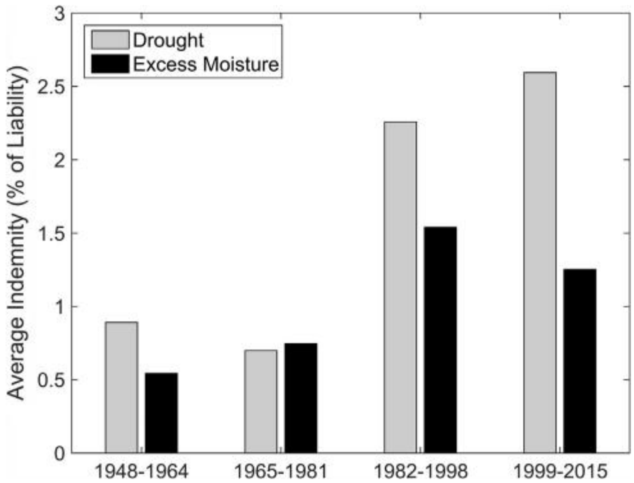

Fig. 9. Average indemnity fraction of liability for two causes of loss – drought and excess moisture – per 17-year intervals in Dane County, Wisconsin.

water table and viewed by the agricultural sector as a non-beneficial ecosystem service – or ecosystem *disservice* (Lyytimaki, 2015; von Dohren and Haase, 2015).

One way in which the agricultural sector has addressed this groundwater flooding issue is to encourage farmer enrollment in federal government conservation programs. An August 2013 news release from the Wisconsin Department of Agriculture, Trade and Consumer Protection suggests that farmers should consider enrolling their flooded fields into the USDA's Conservation Reserve Enhancement Program that pays landowners to return flooded fields to wetlands (WDATCP, 2013). This conservation-focused response contrasts to the historical view of the agricultural community and the federal government that wetlands – often fed by groundwater – should be eliminated by activities such as ditching and tile-draining (Dahl, 1990). However, both views imply a negative relationship between groundwater – and thus groundwater recharge – and agricultural production.

Increased groundwater levels have also likely influenced river flooding in the Upper Yahara River watershed with large peak annual streamflows ($>15\ m^3\ s^{-1}$) increasing in frequency in the last 15 years (Fig. 10). The average annual frequency of these large events nearly doubled from 0.25 for the first half of the record (1976–1981, 1990–1995) to 0.47 for the second half (1996–2014). Although these large events have increased in frequency, the two-sample K–S test shows that the null hypothesis of the early and late period data being from the same distribution cannot be rejected. A coupled groundwater-surface water hydrologic model is needed to fully untangle the influence of groundwater recharge

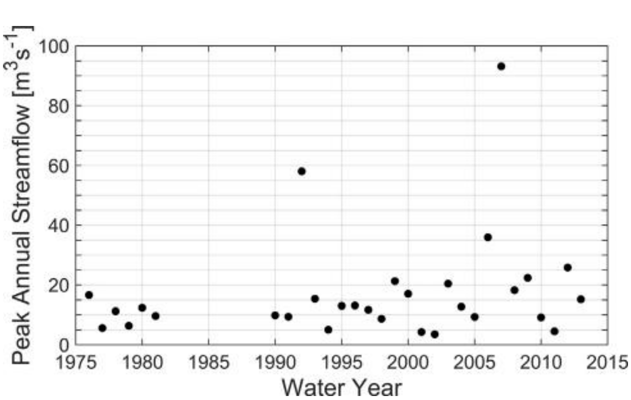

Fig. 10. Peak annual streamflow for the Yahara River at Windsor, Wisconsin USGS gage. The frequency of large flood events ($>15\ m^3\ s^{-1}$) has increased from the first half to the second half of the record.

and precipitation, but elevated groundwater levels reaching the ground surface has been shown to increase the amount of runoff generated from precipitation known as saturation-excess runoff (Dunne and Black, 1970). This impact is likely non-linear and threshold-dependent as more areas become saturated with increasing groundwater levels and thus contribute more runoff (Fig. 1C).

On the other side of the hydrologic spectrum, the issue becomes more complex when considering dry conditions and crops in areas where groundwater recharge can elevate the water table and lead to higher root zone soil moisture and increased crop yield (Soylu et al., 2014; Zipper et al., 2015). Even with a positive precipitation trend, the county encompassing most of the Yahara River watershed has experienced several periods of drought conditions with federal crop insurance indemnity payments for drought losses totaling $14,360,000 (in 2012 dollars) from 1990 to 2011 and an additional $26,497,000 during the extreme 2012 drought (USDA, 2016). Groundwater recharge during those periods would have likely been viewed as a beneficial ecosystem service in support of agricultural production. The overall trend from 1948 to 2015 in the proportion of total crop insurance liabilities that were paid out to farmers as drought-related indemnities has increased even more than that for excess moisture (Fig. 9). While these two trends do not necessarily imply a direct connection to changes in precipitation and recharge, they do indicate the substantial amount of resources spent on mitigating agricultural losses related to excessively wet and dry conditions. The main conclusion is not necessarily that increasing trends in drought- and excess moisture-related impacts exist but that those groundwater-related impacts are substantial and can co-exist.

This conflicting relationship between groundwater recharge and agricultural production is well observed even at the field scale, where both reductions and increases in yield due to the presence of shallow groundwater (groundwater yield penalties and subsidies, respectively) can occur, sometimes even at different times at the same point in the field. Remotely sensed estimates of evapotranspiration from the monitored corn field in the northern part of the watershed during and after a severe drought (May–June 2012) revealed persistent patterns across the field, with certain portions of the field consistently outperforming their surroundings (Zipper and Loheide, 2014). These same patterns are apparent in year-end yield patterns, which indicate that portions of the field with the shallowest groundwater experience yield losses during wet growing season conditions (*groundwater yield penalty*), while sections with intermediate groundwater levels are able to consistently perform at a high level (*groundwater yield subsidy*) and areas with the deepest groundwater consistently produce the worst yield (Zipper et al., 2015).

Fig. 11 shows a simple example of these phenomena using three sites with diverse groundwater levels. In 2012, a year with a severe drought from May to mid-July (red shading), leaf-area index (LAI) is negatively correlated with the groundwater depth (depth to water level; DTWL), indicating that shallow groundwater can help buffer drought stress and enhances crop productivity, thus enhancing an ecosystem service. During 2013, in contrast, two large early-season rain events cause substantial groundwater recharge and saturate the root zone (blue shading) at the shallow site, leading to oxygen stress and reduced LAI relative to the medium groundwater site. However, a lack of rainfall late in the growing season (red shading) once again turns groundwater into a beneficial resource at the shallow groundwater site, allowing it to delay senescence and produce a higher year-end LAI and grain yield (not shown) than the medium groundwater site, thus demonstrating that groundwater recharge can be both an ecosystem disservice and service within the same growing season at the same location.

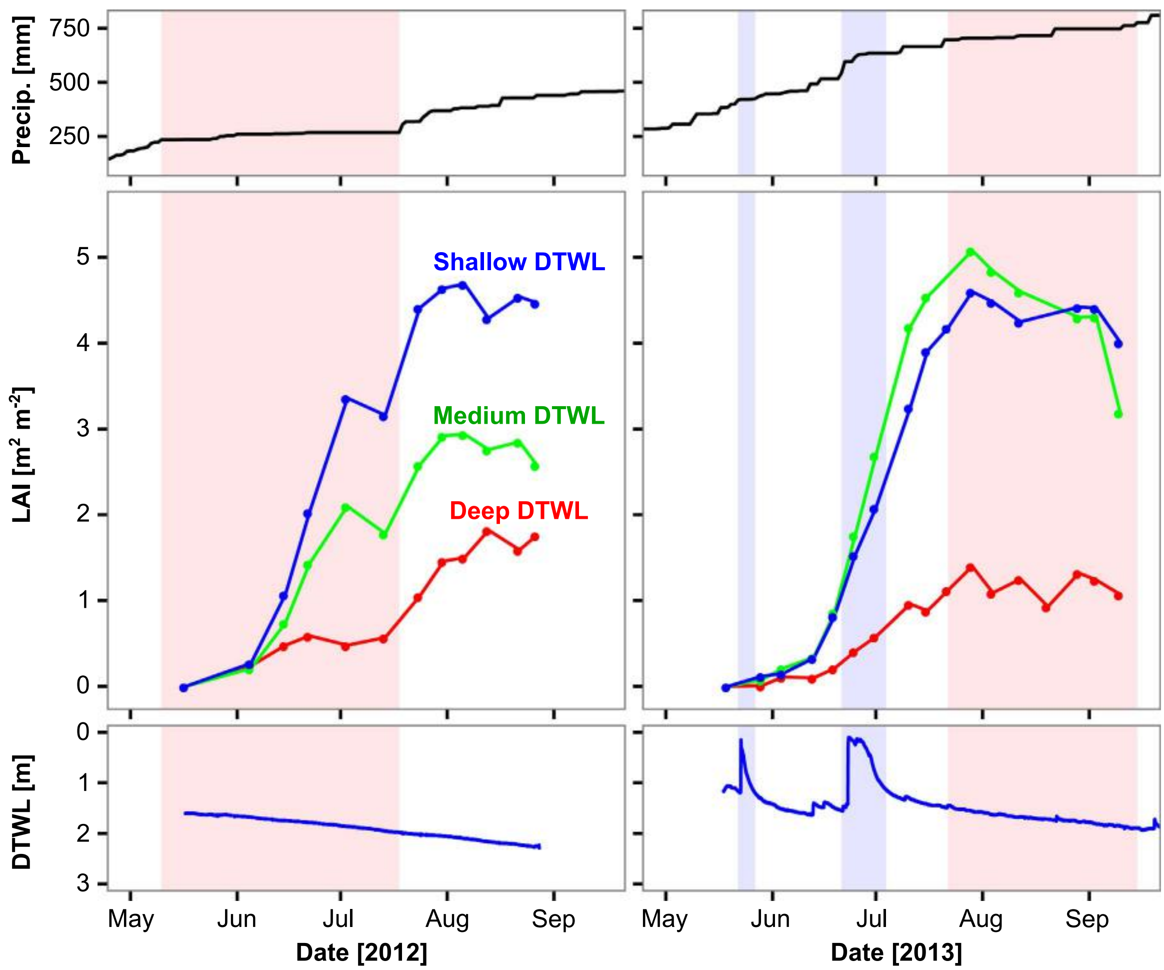

Fig. 11. Paneled plot showing year-to-date precipitation (top), LAI (middle), and DTWL at the shallow site (bottom) for both 2012 (left) and 2013 (right). LAI curves are shown for sites with shallow (0–2.5 m), medium (2–5 m), and deep (5–10 m) DTWL. Red shading in both panels indicates periods of low precipitation where a shallow water table provides a groundwater yield subsidy at the shallow site. Blue shading indicates periods of excessive recharge that lead to oxygen stress and a groundwater yield penalty at the shallow site. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

4. Discussion

Based on the complex and conflicting relationship between groundwater recharge and three final ecosystem services – water supply provision, agricultural production, and flood attenuation – in the Yahara River watershed, we argue that groundwater recharge should be considered only as a hydrologic input into a complex social-ecological system that can both positively and negatively influence human well-being. If water supply to the entire watershed is the only ecosystem service being considered – an uncommon situation – then groundwater recharge could easily be considered an intermediate ecosystem service. However, such an analysis would be of limited use to decision makers interested in specific benefits such as municipal water supply, crop production, and groundwater-dependent ecosystem condition.

In this case, groundwater recharge has low information value as an ecosystem service indicator because of the non-monotonic relationship with two other services (agricultural production and flood attenuation). Therefore, decision makers would benefit from connecting groundwater recharge to final services via mechanistic models as suggested by others (Boyd and Banzhaf, 2007). Agroecosystem and groundwater flow models are capable of transforming biophysical inputs such as groundwater recharge into more relevant variables such as crop productivity (e.g., Soylu et al., 2014) and flows to municipal wells, streams, and wetlands (e.g., Meyer et al., 2014). From these model outputs, other biophysical and ecological models could estimate final ecosystem services such as food produced for human consumption (Cassidy et al., 2013) and wetland habitat condition (Booth and Loheide, 2012).

A disadvantage of only using final ecosystem services such as water supply reliability at a municipal well or wetland habitat condition instead of a service indicator such as groundwater recharge is that the final services cannot be represented continuously across a landscape due to their connection to a specific area or point (Fig. 12). This presents a challenge for integration with other services that are more appropriately mapped across a landscape (e.g., agricultural production or forest recreation). However, the difficulty of mapping certain services reflects the spatial reality of final ecosystem service production and the need to better connect actual production areas or points to final ecosystem service demands and beneficiaries. The allure of a map can certainly draw attention to certain ecosystem processes happening at the landscape-scale. In fact, recharge can still be presented to stakeholders as a hydrologic flux that can potentially be altered due to management interventions to better match a desirable set of final ecosystem services. But it needs to be presented in a more nuanced way that details the positive and negative relationships with multiple services, for example in conjunction with maps of capture area for a municipal well and maps of water table depth in agricultural regions. Ultimately, a map may not always be the best format for accounting for multiple final ecosystem services across a region or watershed.

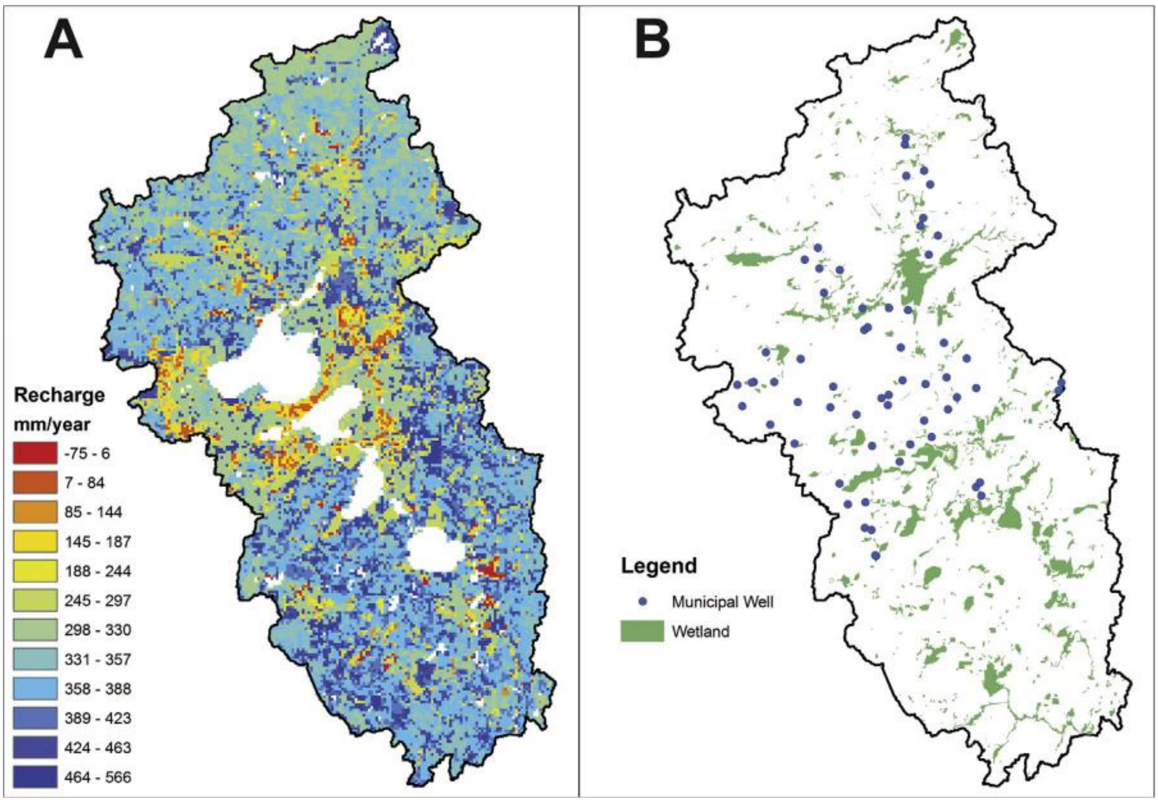

Fig. 12. Maps of the Yahara River watershed showing theoretical values of groundwater recharge across the Yahara River watershed (except open water in white) (A) and locations of wetlands and municipal wells that depend on groundwater supply (B). Recharge can be easily mapped and presented as a water supply provisioning ecosystem service indicator but showing final ecosystem services such as wetland habitat condition or reliability of water supply at municipal wells at specific areas and points allows for better comparison with other final services such as agricultural production.

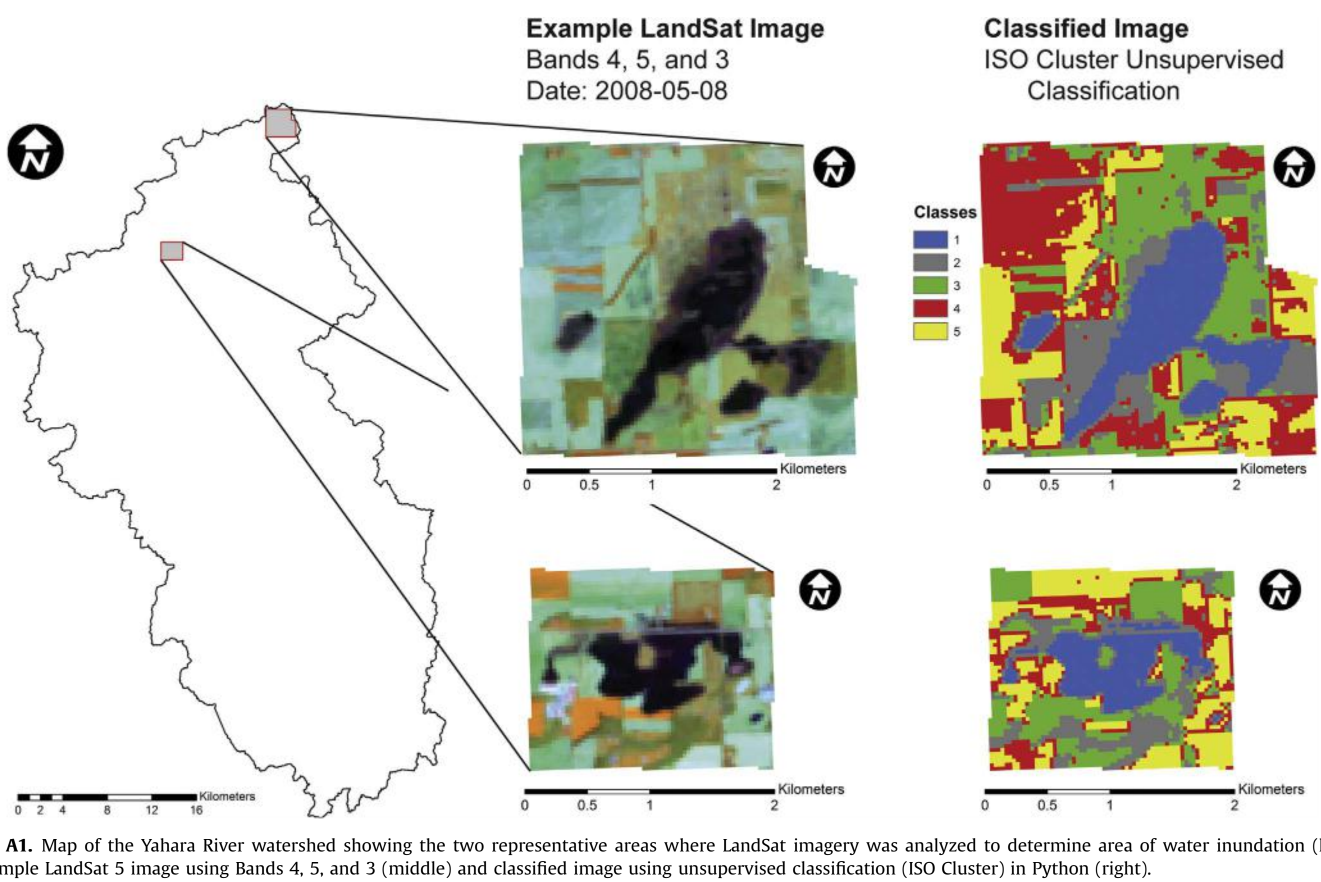

Fig. A1. Map of the Yahara River watershed showing the two representative areas where LandSat imagery was analyzed to determine area of water inundation (left). Example LandSat 5 image using Bands 4, 5, and 3 (middle) and classified image using unsupervised classification (ISO Cluster) in Python (right).

Mechanistic, biophysical models also have the capability of assessing site-specific management interventions that can impact groundwater recharge and associated final ecosystem services. Without such models, mapping groundwater recharge as only a beneficial ecosystem service for water supply provisioning would obscure its role in impacting other services that may be equally important to land managers such as agricultural production and flood attenuation.

Evaluation of ecosystem services has made it clear that tradeoffs are much more common than win-win situations (Carpenter et al., 2009). Therefore, it might seem reasonable to present groundwater recharge and agricultural production together and call their non-monotonic interaction a *tradeoff* in some circumstances and a *synergy* in others. But if a mechanistic relationship (albeit complex and uncertain) is known *a priori*, then we recommend that researchers should attempt to untangle the interactions by creating ecological production function models that can explicitly account for mechanisms when estimating the final ecosystem service value. For example, there is a known mechanistic relationship between recharge and agricultural production through processes such as groundwater flow, water stress, and oxygen stress (Zipper et al., 2015). Our recommendation of exploring these mechanistic relationships is also consistent with Wong et al. (2015) who advocate for using biophysical models to help decision makers connect ecosystem characteristics to final ecosystem services.

Therefore, we argue that it is more appropriate to view recharge as an input to ecological production function models that estimate final services rather than the ecosystem service (or ecosystem service indicator) related to water supply provisioning. Presenting groundwater recharge as an input to the larger social-ecological system that can ultimately provide benefits to humans (i.e. final ecosystem services such as food production, municipal water supply reliability, and wetland habitat) will give clarity to a complex decision-making process.

Using indicators is often a perfectly reasonable and justifiable practice in ecosystem services analysis because we are limited to the current available data and knowledge of the way ecosystems provide benefits to human well-being. However, we must be careful when indicators suggest potentially counter-productive management interventions as when decisions are made based on recharge maps to enhance water supply throughout a watershed that result in reduction in crop yield due to increased plant oxygen stress or increased flood damages. In addition, searching for universal and consistent ecosystem service indicators that can be applied anywhere may be infeasible in some cases due to the inherent spatial variability in climate, geology, soils, and hydrology.

The issue of how to frame groundwater recharge in an ecosystem services context is perhaps less complex in arid regions where groundwater recharge is likely universally beneficial in terms of water supply. In these areas, flooding can occur due to overland flow (infiltration excess, not influenced by high water tables) but groundwater flooding is uncommon. However, dryland regions represent only 40% of the Earth's land area (UNEMG, 2011) and a similar fraction (32%) is influenced by water tables less than 3 m below the surface (Fan et al., 2013). In these relatively wet regions, it cannot be assumed that more water entering the groundwater reservoir via recharge will always lead to beneficial outcomes. Research continues to document cases where high water tables can negatively impact crop production and flooding in regions such as the Pampas in Argentina (Nosetto et al., 2015) and the United Kingdom (Hughes et al., 2011). Some of these wet regions are likely to be impacted by wetter climates in the future (IPCC, 2014) where increased groundwater recharge may exacerbate problems further. In addition, salt mobilization associated with groundwater recharge is a well-documented water quality problem in some arid regions such as Australia and California (Noorduijn et al., 2010) where groundwater recharge could be framed as negatively impacting agriculture and water quality.

A fundamental concept in water management is to manage for extremes; flood management deals with too much water and water supply management deals with not enough water. Thus, the challenge is to moderate the extremes while protecting ecosystems that have evolved within those extremes. Effective solutions must address both the wet and dry extremes. For example, the management of a water reservoir must balance having storage available for an incoming flood event to minimize downstream flood damages with having enough water stored in the event of a drought to minimize downstream water supply shortages. Thus, we recommend that hydrologic services such as groundwater recharge be framed within this spectrum of hydrologic extremes and treated as processes – rather than exclusively beneficial services – within a complex social-ecological system that can ultimately influence costs and benefits to human well-being.

5. Conclusion

We present a case study with global implications that reveals a complex relationship between a common hydrologic ecosystem service indicator – groundwater recharge – and final ecosystem services such as water supply provisioning, agricultural production, and flood attenuation. While recharge is almost universally framed as a beneficial service as it increases groundwater supplies by its very definition, it can also impact other final ecosystem services in non-linear ways by: 1) both positively and negatively impacting agricultural production through supplying crops with moisture during times of water stress and inundating the root zones of crops leading to oxygen stress, respectively (Fig. 1B); and 2) increasing the risk of flooding due to elevated water tables and increasing runoff response (Fig. 1C).

During a time when the popularity of ecosystem service assessments and particularly payments for ecosystem services schemes are increasing globally, it behooves the ecological and hydrologic science community to strongly consider how hydrologic fluxes are treated in such frameworks as others have suggested (Guswa et al., 2014). We should encourage the use and further development of biophysical models and ecological production functions – as others have advocated (Wong et al., 2015) – to move beyond potential intermediate ecosystem services and towards final services such as water supply reliability at a well location. As this study suggests, more clarity regarding the complex, non-monotonic behavior of hydrologic processes in supporting or hindering final ecosystem services needs to be highlighted in the ecosystem service science community. We also recommend that ecosystem service mapping activities should explicitly separate intermediate and final services when aggregating multiple services together to determine tradeoffs, synergies, and win-win situations.

Acknowledgments

We would like to thank Erin Gross for collecting information on local stormwater regulations, Cheryl Buchwald and Faith Fitzpatrick of the U.S. Geological Survey's Wisconsin Water Science Center for supplying groundwater withdrawal and historical spring flow data, and Dave Taylor of the Madison Metropolitan Sewage Destrict for providing wastewater effluent discharge data. We would also like to thank Jiangxiao Qiu for providing helpful comments on an earlier version of this manuscript. The manuscript has been substantially improved thanks to valuable comments from four anonymous reviewers. This material is based upon work supported by the National Science Foundation under grant DEB-1038759 – Water Sustainability and Climate and under Cooperative Agreement #DEB-1440297, North Temperate Lakes Long Term Ecological Research.

Appendix A. Calculating inundation area using LandSat

Clear-sky LandSat 5 Thematic Mapper images of two representative areas in the northern part of the Yahara watershed (Fig. A1) from 1984 to 2011 were downloaded from the U.S. Geological Survey's Global Visualization Viewer (http://glovis.usgs.gov/). These areas interact with shallow groundwater and thus area of water inundation is a useful indicator of groundwater levels although this relationship is not linear due to topographic variability. The Python programming language was used for all image post-processing. Composite images of Bands 3, 4, and 5 were created for each date. This combination has been shown to highlight the contrast between land and water, which is critical for identifying inundation area. Unsupervised classification (ISO Cluster) was used to then classify each image into 5 classes (Fig. A1), of which the first was always open water. The number of pixels classified as open water was then counted and converted to area.

References

Baker, J.M., Griffis, T.J., Ochsner, T.E., 2012. Coupling landscape water storage and supplemental irrigation to increase productivity and improve environmental stewardship in the U.S. Midwest. Water Resour. Res., 48.
Baral, H., Keenan, R.J., Fox, J.C., Stork, N.E., Kasel, S., 2013. Spatial assessment of ecosystem goods and services in complex production landscapes: a case study from south-eastern Australia. Ecol. Complex. 13, 35–45.
Barbier, E.B., 2007. Valuing ecosystem services as productive inputs. Econ. Policy, 178–229.
Barlow, P.M., Leake, S.A., 2012. Streamflow depletion by wells—Understanding and managing the effects of groundwater pumping on streamflow: U.S. Geological Survey Circular 1376, p. 84.
Baumann, R., 1968. Cherokee Marsh: Win, Draw, Compromise? A Study of Democracy in Action. Institute of Governmental Affairs, University Extension, University of Wisconsin,, Madison, Wisconsin.
Bennett, E.M., Peterson, G.D., Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. Ecol. Lett. 12, 1394–1404.
Berbés-Blázquez, M., González, J.A., Pascual, U., 2016. Towards an ecosystem services approach that addresses social power relations. Curr. Opin. Environ. Sustain 19, 134–143.
Booth, E.G., Loheide II, S.P., 2012. Hydroecological model predictions indicate wetter and more diverse soil water regimes and vegetation types following floodplain restoration. J. Geophys. Res. 117, G02011.
Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ. 63, 616–626.
Bradbury, K.R., Swanson, S.K., Krohelski, J.T., Fritz, A.K., 1999. Hydrogeology of Dane County, Wisconsin Open-File Report 1999-04. Wisconsin Geological and Natural History Survey,, Madison, WI, p. 68.
Brauman, K.A., 2015. Hydrologic ecosystem services: linking ecohydrologic processes to human well-being in water research and watershed management. Wiley Interdiscip. Rev. Water 2, 345–358.
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.
Burkhard, B., Kroll, F., Nedkov, S., Muller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Indic. 21, 17–29.
Burkhard, B., Crossman, N., Nedkov, S., Petz, K., Alkemade, R., 2013. Mapping and modelling ecosystem services for science, policy and practice. Ecosyst. Serv. 4, 1–3.
Butler, C.D., Oluoch-Kosura, W., 2006. Linking future ecosystem services and future human well-being. Ecol. Soc., 11.
Cadavid, C.L., Ando, A.W., 2013. Valuing preferences over stormwater management outcomes including improved hydrologic function. Water Resour. Res. 49, 4114–4125.
Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., DeFries, R.S., Diaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., Sarukhan, J., Scholes, R.J., Whyte, A., 2009. Science for managing ecosystem services: Beyond the millennium ecosystem assessment. Proc. Natl. Acad. Sci. USA 106, 1305–1312.
Cassidy, E.S., West, P.C., Gerber, J.S., Foley, J.A., 2013. Redefining agricultural yields: from tonnes to people nourished per hectare. Environ. Res. Lett. 8.
Castro, A.J., Martín-López, B., López, E., Plieninger, T., Alcaraz-Segura, D., Vaughn, C. C., Cabello, J., 2015. Do protected areas networks ensure the supply of ecosystem services? Spatial patterns of two nature reserve systems in semi-arid Spain. Appl. Geogr. 60, 1–9.
Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. Biol. Conserv. 141, 350–352.
Costanza, R., dArge, R., deGroot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., ONeill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., vandenBelt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.
Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E. G., Martín-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B., Maes, J., 2013. A blueprint for mapping and modelling ecosystem services. Ecosyst. Serv. 4, 4–14.
Dahl, T.E., 1990. Wetland losses in the United States, 1780s to 1980s. U.S. Fish and Wildlife Service,, Washington, DC.
Daily, G.C., 1997. Nature's Services. Island Press,, Covelo, CA.
Dempsey, J., Robertson, M.M., 2012. Ecosystem services: tensions, impurities, and points of engagement within neoliberalism. Prog. Hum. Geogr. 36, 758–779.
Dripps, W.R., Bradbury, K.R., 2007. A simple daily soil-water balance model for estimating the spatial and temporal distribution of groundwater recharge in temperate humid areas. Hydrogeol. J. 15, 433–444.
Dunne, T., Black, R.D., 1970. Partial area contributions to storm runoff in a small new-england watershed. Water Resour. Res. 6, 1296-&.
Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. Agric. Ecosyst. Environ. 127, 135–140.
Ernstson, H., Sorlin, S., 2013. Ecosystem services as technology of globalization: On articulating values in urban nature. Ecol. Econ. 86, 274–284.
Euliss, N.H., Smith, L.M., Wilcox, D.A., Brwaine, B.A., 2008. Linking ecosystem processes with wetland management goals: charting a course for a sustainable future. Wetlands 28, 553–562.
Fan, Y., Li, H., Miguez-Macho, G., 2013. Global Patterns of groundwater table depth. Science 339, 940–943.
Ferguson, B.K., 1990. Role of the long-term water-balance in management of stormwater infiltration. J. Environ. Manag. 30, 221–233.
Fetter, C.W., 1977. Statistical analysis of impact of ground-water pumpage on low-flow hydrology. Water Resour. Bull. 13, 309–323.
Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643–653.
Garrick, D., Siebentritt, M.A., Aylward, B., Bauer, C.J., Purkey, A., 2009. Water markets and freshwater ecosystem services: policy reform and implementation in the columbia and Murray-Darling basins. Ecol. Econ. 69, 366–379.
Gillon, S., Booth, E.G., Rissman, A.R., 2015. Shifting drivers and static baselines in environmental governance: challenges for improving and proving water quality outcomes. Reg. Environ. Chang., 1–17.
Gleeson, T., Wada, Y., Bierkens, M.F.P., van Beek, L.P.H., 2012. Water balance of global aquifers revealed by groundwater footprint. Nature 488, 197–200.
Gotkowitz, M.B., Attig, J.W., McDermott, T., 2014. Groundwater flood of a river terrace in southwest Wisconsin, USA. Hydrogeol. J. 22, 1421–1432.
Guswa, A.J., Brauman, K.A., Brown, C., Hamel, P., Keeler, B.L., Sayre, S.S., 2014. Ecosystem services: challenges and opportunities for hydrologic modeling to support decision making. Water Resour. Res. 50, 4535–4544.
Hamel, P., Daly, E., Fletcher, T.D., 2013. Source-control stormwater management for mitigating the impacts of urbanisation on baseflow: a review. J. Hydrol. 485, 201–211.
Hart, D.J., Schoephoester, P.R., Bradbury, K.R., 2012. Groundwater recharge in Dane County, Wisconsin: estimating recharge using a GIS-based water-balance model, Bulletin 107. Wisconsin Geological and Natural History Survey,, Madison, Wisconsin.
Holman, I.P., 2006. Climate change impacts on groundwater recharge-uncertainty, shortcomings, and the way forward? Hydrogeol. J. 14, 637–647.
Holman, I.P., Tascone, D., Hess, T.M., 2009. A comparison of stochastic and deterministic downscaling methods for modelling potential groundwater recharge under climate change in East Anglia, UK: implications for groundwater resource management. Hydrogeol. J. 17, 1629–1641.
Hughes, A.G., Vounaki, T., Peach, D.W., Ireson, A.M., Jackson, C.R., Butler, A.P., Bloomfield, J.P., Finch, J., Wheater, H.S., 2011. Flood risk from groundwater: examples from a Chalk catchment in southern England. J. Flood Risk Manag. 4, 143–155.
Hunt, R.J., Steuer, J.J., Mansor, M.T.C., Bullen, T.D., 2001. Delineating a recharge area for a spring using numerical modeling, Monte Carlo techniques, and geochemical investigation. Ground Water 39, 702–712.
IPCC, 2014. Barros, V.R., Field, C.B., Dokken, D.J., Mastrandrea, M.D., Mach, K.J., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Johnston, R.J., Russell, M., 2011. An operational structure for clarity in ecosystem service values. Ecol. Econ. 70, 2243–2249.

Jones, M., Jones, J., Olsen, T., 2000. Ground-Water Flooding in Glacial Terrain of Southern Puget Sound, Washington, Fact Sheet 111-00. U.S. Geological Survey,, Tacoma, Washington.
Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S., 2011. Natural Capital - Theory and Practice of Mapping Ecosystem Services. Oxford University Press,, New York.
Kinzig, A.P., Perrings, C., Chapin, F.S., Polasky, S., Smith, V.K., Tilman, D., Turner, B.L., 2011. Paying for ecosystem services-promise and peril. Science 334, 603–604.
Kreibich, H., Thieken, A.H., 2008. Assessment of damage caused by high groundwater inundation. Water Resour. Res., 44.
Krohelski, J.T., Bradbury, K.R., Hunt, R.J., Swanson, S.K., 2000. Numerical Simulation of Groundwater Flow in Dane County, Wisconsin, Bulletin 98. Wisconsin Geological and Natural History Survey,, Madison, Wisconsin, p. 31.
Kronenberg, J., Hubacek, K., 2013. Could payments for ecosystem services create an "Ecosystem Service Curse"? Ecol. Soc., 18.
Krug, W.R., 1999. Simulation of the effects of operating lakes Mendota, Monona, and Waubesa, south-central Wisconsin, as multipurpose reservoirs to maintain dry-weather flow, Open-File Report, - ed. U.S. Dept. of the Interior, U.S. Geological Survey ; Branch of Information Services [distributor], pp. iv, 18 p. ill., map ;28 cm.
Kucharik, C.J., Serbin, S.P., Vavrus, S., Hopkins, E.J., Motew, M.M., 2010. Patterns of climate change across Wisconsin from 1950 to 2006. Phys. Geogr. 31, 1–28.
Kuppel, S., Houspanossian, J., Nosetto, M.D., Jobbagy, E.G., 2015. What does it take to flood the Pampas? Lessons from a decade of strong hydrological fluctuations. Water Resour. Res. 51, 2937–2950.
Liu, J.G., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C.L., Schneider, S.H., Taylor, W.W., 2007. Complexity of coupled human and natural systems. Science 317, 1513–1516.
Livick, B., 2013. Village: Flood control measures have paid off, Oregon Observer, Oregon, Wisconsin.
Lowry, C.S., Loheide, S.P., 2010. Groundwater-dependent vegetation: Quantifying the groundwater subsidy. Water Resour. Res. 46.
Lyytimaki, J., 2015. Ecosystem disservices: embrace the catchword. Ecosyst. Serv. 12, 136-136.
Mace, G.M., Baillie, J.E.M., 2007. The 2010 biodiversity indicators: challenges for science and policy. Conserv. Biol. 21, 1406–1413.
Macholl, J.A., 2007. Inventory of Wisconsin's Springs, Open-File Report 2007-03. Wisconsin Geological and Natural History Survey,, Madison, WI.
McLeod, R.S., 1978. Water-Level Declines in the Madison Area, Dane County, Wisconsin, Open-File Report 78–936, - ed. U.S. Geological Survey, Madison, Wisconsin.
MEA, 2005. Millenium Ecosystem Assessment, Ecosystems and Human Well-being: Synthesis. Island Press,, Washington, DC.
Meyer, P.A., Brouwers, M., Martin, P.J., 2014. A three-dimensional groundwater flow model of the Waterloo Moraine for water resource management. Can. Water Resour. J. 39, 167–180.
Motew, M.M., Kucharik, C.J., 2013. Climate-induced changes in biome distribution, NPP, and hydrology in the Upper Midwest U.S.: a case study for potential vegetation. J. Geophys. Res. Biogeosci. 118, 248–264.
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.
Noorduijn, S.L., Ghadouani, A., Vogwill, R., Smettem, K.R.J., Legendre, P., 2010. Water table response to an experimental alley farming trial: dissecting the spatial and temporal structure of the data. Ecol. Appl. 20, 1704–1720.
Nosetto, M.D., Jobbagy, E.G., Jackson, R.B., Sznaider, G.A., 2009. Reciprocal influence of crops and shallow ground water in sandy landscapes of the Inland Pampas. Field Crop. Res. 113, 138–148.
Nosetto, M.D., Paez, R.A., Ballesteros, S.I., Jobbágy, E.G., 2015. Higher water-table levels and flooding risk under grain vs. livestock production systems in the subhumid plains of the Pampas. Agric. Ecosyst. Environ. 206, 60–70.
Olander, L., Johnston, R.J., Tallis, H., Kagan, J., Maguire, L., Polasky, S., Urban, D., Boyd, J., Wainger, L., Palmer, M., 2015. Best Practices for Integrating Ecosystem Services into Federal Decision Making. National Ecosystem Services Partnership, Duke University,, Durham, NC.
Polasky, S., Tallis, H., Reyers, B., 2015. Setting the bar: standards for ecosystem services. Proc. Natl. Acad. Sci. USA 112, 7356–7361.
Postel, S.L., Daily, G.C., Ehrlich, P.R., 1996. Human appropriation of renewable fresh water. Science 271, 785–788.
Potter, K.W., 1991. Hydrological impacts of changing land management-practices in a moderate-sized agricultural catchment. Water Resour. Res. 27, 845–855.
Potter, K.W., 2006. Small-scale, spatially distributed water management practices: Implications for research in the hydrologic sciences. Water Resour. Res., 42.
Power, A.G., 2010. Ecosystem services and agriculture: tradeoffs and synergies. Philos. Trans. R. Soc. Lond. B Biol. Sci. 365, 2959–2971.
Pryor, S.C., Howe, J.A., Kunkel, K.E., 2009. How spatially coherent and statistically robust are temporal changes in extreme precipitation in the contiguous USA? Int. J. Clim. 29, 31–45.
Qian, T., Dai, A., Trenberth, K.E., 2007. Hydroclimatic trends in the Mississippi River basin from 1948 to 2004. J. Clim. 20, 4599–4614.
Qiu, J., Turner, M.G., 2013. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proc. Natl. Acad. Sci. USA 110, 12149–12154.
Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. USA 107, 5242–5247.
Ren, S.Q., Kingsford, R.T., 2014. Modelling impacts of regulation on flows to the Lowbidgee floodplain of the Murrumbidgee River, Australia. J. Hydrol. 519, 1660–1667.
Reyers, B., Biggs, R., Cumming, G.S., Elmqvist, T., Hejnowicz, A.P., Polasky, S., 2013. Getting the measure of ecosystem services: a social-ecological approach. Front. Ecol. Environ. 11, 268–273.
Rickert, C., 2008. Storm Long Gone, but the Water Hangs Around. Wisconsin State Journal,, Madison, Wisconsin, p. D1.
Ringold, P.L., Nahlik, A.M., Boyd, J.W., Bernard, D., 2011. Report from the Workshop on Indicators of Final Ecosystem Goods and Services for Wetlands and Estuaries. U.S. Environmental Protection Agency, Denver, Colorado, p. 73.
Ringold, P.L., Boyd, J., Landers, D., Weber, M., 2013. What data should we collect? A framework for identifying indicators of ecosystem contributions to human well-being. Front. Ecol. Environ. 11, 98–105.
Robertson, M., BenDor, T.K., Lave, R., Riggsbee, A., Ruhl, J.B., Doyle, M., 2014. Stacking ecosystem services. Front. Ecol. Environ. 12, 186–193.
Roy, A.H., Wenger, S.J., Fletcher, T.D., Walsh, C.J., Ladson, A.R., Shuster, W.D., Thurston, H.W., Brown, R.R., 2008. Impediments and solutions to sustainable, watershed-scale urban stormwater management: Lessons from Australia and the United States. Environ. Manag. 42, 344–359.
Schaefer, M., Goldman, E., Bartuska, A.M., Sutton-Grier, A., Lubchenco, J., 2015. Nature as capital: Advancing and incorporating ecosystem services in United States federal policies and programs. Proc. Natl. Acad. Sci. USA 112, 7383–7389.
Serna-Chavez, H.M., Schulp, C.J.E., van Bodegom, P.M., Bouten, W., Verburg, P.H., Davidson, M.D., 2014. A quantitative framework for assessing spatial flows of ecosystem services. Ecol. Indic. 39, 24–33.
Shackleton, C.M., Ruwanza, S., Sinasson Sanni, G.K., Bennett, S., De Lacy, P., Modipa, R., Mtati, N., Sachikonye, M., Thondhlana, G., 2016. Unpacking pandora's box: understanding and categorising ecosystem disservices for environmental management and human wellbeing. Ecosystems 19, 587–600.
Simms, P., Leaf, N., 2007. Claims Pouring in; Basements Likely Not Covered. Wisconsin State Journal,, Madison, Wisconsin, p. A5.
Soren, J., 1976. Basement Flooding and Foundation Damage From Water-Table Rise in the East New York Section of Brooklyn, Long Island, New York, Water-Resources Investigations 76-95. U.S. Geological Survey,, Mineola, New York.
Soylu, M.E., Kucharik, C.J., Loheide, S.P., 2014. Influence of groundwater on plant water use and productivity: development of an integrated ecosystem - Variably saturated soil water flow model. Agric. . Meteorol. 189, 198–210.
Swanson, S.K., Bradbury, K.R., Hart, D.J., 2009. Assessing the vulnerability of spring systems to groundwater withdrawals in southern Wisconsin. Geosci. Wis., 20.
Turner, R.K., Daily, G.C., 2008. The ecosystem services framework and natural capital conservation. Envir. Resour. Econ. 39, 25–35.
UNEMG, 2011. Global Drylands: A UN system-wide response. United Nations Environment Management Group.
USDA, 2016. United States Department of Agriculture, Risk Management Agency, Cause of Loss Historical Data Files. http://www.rma.usda.gov/data/cause.html.
Vigerstol, K.L., Aukema, J.E., 2011. A comparison of tools for modeling freshwater ecosystem services. J. Environ. Manag. 92, 2403–2409.
Vogel, R.M., Lall, U., Cai, X.M., Rajagopalan, B., Weiskel, P.K., Hooper, R.P., Matalas, N. C., 2015. Hydrology: The interdisciplinary science of water. Water Resour. Res. 51, 4409–4430.
von Dohren, P., Haase, D., 2015. Ecosystem disservices research: a review of the state of the art with a focus on cities. Ecol. Indic. 52, 490–497.
Wada, Y., van Beek, L.P.H., van Kempen, C.M., Reckman, J.W.T.M., Vasak, S., Bierkens, M.F.P., 2010. Global depletion of groundwater resources. Geophys. Res. Lett., 37.
Wallace, K.J., 2007. Classification of ecosystem services: problems and solutions. Biol. Conserv. 139, 235–246.
WDATCP, 2013. Flooded Fields Again This Year? Think About CREP. In: Proceedings of the Wisconsin Department of Agriculture, T.a.C.P. (Ed.), Madison, Wisconsin.
Wegener, M.W., 2001. Long-term land use/cover change patterns in the Yahara Lakes region and their impact on runoff volume to Lake Mendota, Environmental Monitoring. University of Wisconsin-Madison,, Madison, WI.
Wong, C.P., Jiang, B., Kinzig, A.P., Lee, K.N., Ouyang, Z.Y., 2015. Linking ecosystem characteristics to final ecosystem services for public policy. Ecol. Lett. 18, 108–118.
Young, K.B., 1966. Effect of treated effluent diversion on Yahara River flow, Wisconsin, Open-File Report 66-0157, - ed. U.S. Geological Survey.
Zipper, S.C., Loheide, S.P., 2014. Using evapotranspiration to assess drought sensitivity on a subfield scale with HRMET, a high resolution surface energy balance model. Agric. Meteorol. 197, 91–102.
Zipper, S.C., Soylu, M.E., Booth, E.G., Loheide, S.P., 2015. Untangling the effects of shallow groundwater and soil texture as drivers of subfield-scale yield variability. Water Resour. Res. 51, 6338–6358.