Assessing Impacts of Payments for Watershed Services on Sustainability in Coupled Human and Natural Systems

HEIDI ASBJORNSEN, ALEX S. MAYER, KELLY W. JONES, THERESA SELFA, LEONARDO SAENZ, RANDALL K. KOLKA, AND KATHLEEN E. HALVORSEN

Payments for watershed services (PWS) as a policy tool for enhancing water quality and supply have gained momentum in recent years, but their ability to lead to sustainable watershed outcomes is uncertain. Consequently, the demand for effective monitoring and evaluation (M&E) of PWS impacts on coupled human and natural systems (CHANS) and their implications for watershed sustainability (WS) is increasing. The theoretical foundations and practical applications of WS frameworks, which integrate biophysical and socioeconomic indicators to assess progress toward WS goals, have been extensively explored but rarely applied to PWS. We develop the PWS–WS framework as an approach for guiding indicator selection to improve knowledge about the complex drivers, interactions, and feedback between PWS and CHANS. A review of the PWS and WS literatures provides a basis for comparing and contrasting indicators. Using two case studies, we illustrate how applying the PWS–WS framework using a place-based, contextualized approach enhances potential for sustainable watershed outcomes.

Keywords: payments for ecosystem services, water quality and quantity, indicators, monitoring and evaluation, interdisciplinary research

Hydrologic services are arguably one of the most crucial and threatened ecosystem services for sustaining human societies. With the growing problems of water scarcity and declining water quality, payments for watershed services (PWS) have surged worldwide in recent years (Goldman-Benner et al. 2012). By connecting hydrologic service suppliers and consumers in ways that explicitly incorporate a market value for such services, PWS seek to eliminate the externalities distorting traditional economic markets and to create incentives for conservation that are equal to or greater than the opportunity costs foregone by limiting land-use options. If they are successful, such approaches could vastly improve the supply and quality of water resources, especially in areas experiencing threats of declining watershed services.

Despite the laudable goals and growing popularity of PWS worldwide, they have received considerable criticism in recent years related to their failure to adequately document progress toward achieving the targeted hydrologic outcomes (Locatelli and Vignola 2009, Brouwer et al. 2011), as well as for indirect effects that have led to undesirable social, economic, and environmental consequences (Daw et al. 2011, Goldman-Benner et al. 2012, Shapiro-Garza 2013). These shortcomings reflect a fundamental lack of understanding of the complex interactions and feedback occurring between market-based approaches such as PWS and the coupled human and natural systems (CHANS) in which they operate (Shapiro-Garza 2013). Unless effective accounting for these complexities is incorporated into PWS design and evaluation, such water markets will likely fail in achieving desired long-term impacts on watershed sustainability.

Insufficient or absent monitoring and evaluation (M&E) of PWS performance is commonly cited as a primary limitation to identifying both their direct and indirect impacts on CHANS—information that is crucial to evaluating progress toward stated goals and adapting program activities to maximize their benefits and minimize undesirable consequences. These observations have led to a strong call for more effective M&E to advance the understanding of PWS–CHANS relationships, thereby providing a basis for improving PWS policies to better maximize benefit–cost trade-offs and

BioScience 65: 579–591.
doi:10.1093/biosci/biv051

sustainability within watersheds (Jack et al. 2008, Brouwer et al. 2011, Porras et al. 2013).

To be effective, M&E programs should incorporate a set of indicators that are relatively easy to measure, have a high degree of sensitivity to the conditions of interest, and provide reliable information over time about the achievement of desired outcomes (Dale and Beyeler 2001, Reyers et al. 2013). Several fundamental characteristics distinguish PWS from other policy instruments and require consideration when developing M&E programs: First, PWS are designed to operate within well-defined watershed boundaries so that the feedback and interactions among biophysical and socioeconomic components generally occur within a common geographic spatial unit. Second, a primary focus of PWS programs is to create links between land-use decisions by upstream water producers and the quality and quantity of hydrologic services available to downstream water consumers through the use of market transactions (e.g., the exchange of funds and services between city dwellers and rural landowners) and by influencing knowledge and perceptions (e.g., about forest–water relationships and the impacts of land-use change on water resources) (Kosoy et al. 2007). Third, PWS are primarily concerned with changing human behavior specifically related to land-use practices in ways that positively affect water resources (e.g., setting aside forests or adopting favorable land-use practices); this is in contrast to other incentive-based policies that target behaviors related to water usage (e.g., reducing consumption) or market-based cap-and-trade approaches. Finally, PWS programs generally seek to achieve long-term sustainable provisioning of watershed benefits, reflected by multiple-year contracts; efforts to generate complementary funding sources to ensure continued support; and investments in educational, training, and other-capacity building initiatives intended to have long-lasting effects.

Although rarely invoked explicitly within a PWS context, the concept of *watershed sustainability* (WS) encompasses many of the unique aspects of the PWS programs described above, providing opportunities to learn from and adapt its extensive experience with using indicators to monitor and evaluate progress. Broadly defined as "the use of water that supports the ability of human society to endure and flourish into the indefinite future without undermining the integrity of the hydrological cycle or the ecological systems that depend on it" (Gleick 1998), the concept of WS has evolved over centuries. For example, Neary (2000) noted several civilizations that recognized the need to properly manage watersheds, including the Vedic (approximately 800 BCE in today's India), the Hohokam (approximately 1000 CE in today's Arizona), and medieval French (approximately 1200 CE) cultures. Chorley (1969) credited the British with first using watershed boundaries for planning and administration in 1752. John Wesley Powell is credited with developing the first large-scale watershed management plan in 1890, for the western United States (Kenney 1999). More recently, the concept of *watershed management* builds on objectives that reflect the complex socioecological processes of CHANS that determine WS, as summarized by Kneese (1964): (a) determine the desirable quantity and quality of water to be maintained, (b) devise the best biophysical system for achieving that quantity and quality, and (c) determine the best institutional arrangements for administering and managing water quality and quantity. In the context of PWS, the first process relates to identifying indicators of the state of watershed services, the second process to land use and other human activities affecting quality and quantity, and the third process to the role of governance in managing water resources. Similarly, the conceptual framework of *integrated watershed resources management*, popularized since the 1980s, includes as significant components the interactions between upstream and downstream actions and the role of diverse actors in watersheds (Heinz et al. 2007). The concept of *sociohydrology* focuses on modeling the interactions and, especially, the feedback between the biophysical and socioeconomic dimensions in watersheds (e.g., Sivapalan et al. 2011, Gober 2014). Other concepts related to WS include *water security*, defined as the sustainable access to adequate quantities of water of acceptable quality to ensure human and ecosystem health (Norman et al 2013); the *water poverty index*, an assessment of water scarcity that takes into account physical estimates of water availability and the socioeconomic drivers of poverty (Sullivan 2002); the *enhanced water poverty index*, which integrates the concept of causality in the water poverty index through the pressure–state–response model (Perez-Foguet and Garriga 2011); and *water resource vulnerability*, the susceptibility of a system (individual, community, place) to be damaged as a function of exposure to external forces (shocks, stress, disturbances), sensitivity of the system, and the ability of the system to respond (cope, recover, adapt) (Plummer et al. 2012). Indicators for assessing WS have been applied to a wide range of management, environmental, and social contexts (e.g., Karr 1991, Jimenez-Cisneros 1996, Chaves and Alipaz 2007, Juwana et al. 2010, Yoon et al. 2014); however, a common salient theme is the integration of indicators from the human and biophysical systems to enhance understanding of the overall condition of the watershed. The basic premise of PWS—to foster interactions that support the provisioning of hydrologic services, which, in turn, are dependent on the social well-being and economic vitality of both water providers and users—fits squarely within the WS concept. As we illustrate in figure 1, PWS can be conceptualized through the sociohydrology lens as a series of interactions and feedback between policies, the decisions and actions of individuals and societies, and the state of hydrologic services, all of which need to be effectively captured by indicators to assess progress toward WS.

Despite numerous calls for improving PWS M&E to address the lack of documentation of their performance and maximize their potential for securing water resources for society, to date, no comprehensive PWS-specific M&E framework exists. Our objectives for this article are (a) to develop an

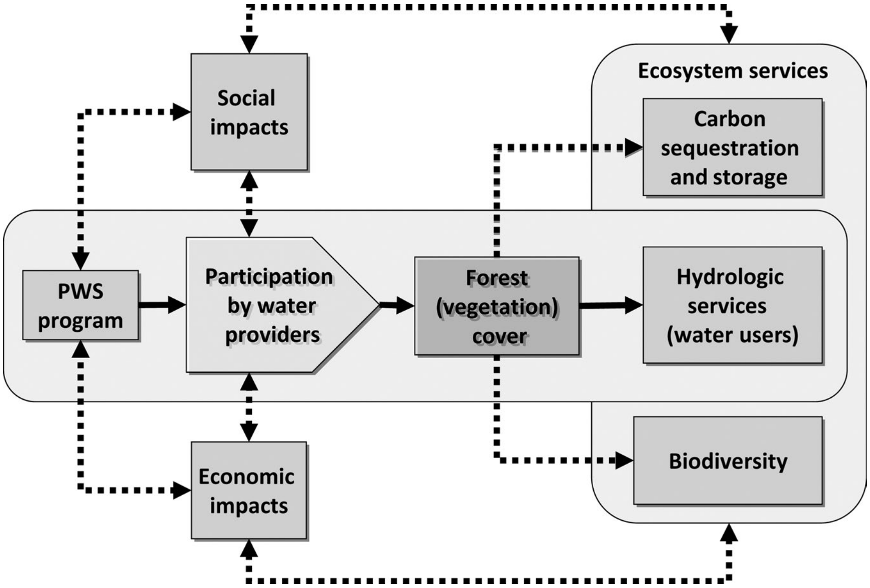

Figure 1. A conceptual model showing how payment for watershed services (PWS) programs typically operate within coupled human and natural systems (CHANS): They are designed to have direct impacts on forest cover and (presumably) hydrologic services but often include indirect impacts on other ecosystem services and on the social and economic systems while generating (often unexpected) feedback between the different components of the CHANS.

integrated *PWS–WS framework* that explicitly acknowledges the CHANS in which PWS operate and incorporates the social and environmental dimensions that underpin WS; (b) on the basis of a review of the current theory and practice, to compare and contrast approaches for selecting meaningful M&E indicators for assessing progress of PWS toward WS and incorporate the results into our PWS–WS framework; and (c) to apply the PWS–WS framework to PWS case studies to illustrate the opportunities and constraints of this approach for advancing fundamental knowledge about CHANS dynamics in response to PWS and enhancing the effectiveness of PWS to achieve WS outcomes. Our analysis is distinct from several recent reviews of PWS (Pagiola 2008, Lele 2009, Brouwer et al. 2011, Whittington and Porras et al. 2013) because it explicitly integrates CHANS and WS concepts within PWS theory and practice, synthesizes the state of M&E indicators in the PWS and WS literature, and offers a PWS-specific framework firmly grounded in previous watershed management initiatives but having broad application to PWS M&E.

Coupled human and natural systems and PWS: The PWS–WS framework

The underlying theory and concepts of CHANS build on the idea that complex interactions and feedback between societies and the environment will affect system properties, such as the degree of stability (i.e., the ability to maintain the ecosystem's structure and functions) and resilience to change (i.e., the recovery of the ecosystem's structure and functions following a disruption), threshold dynamics (e.g., tipping points that determine shifts in ecosystem state), legacy effects and time lags, and emergent properties (Liu et al. 2007), all of which have important implications for the sustainability of CHANS (Ostrom 2009). PWS inherently operate at the interface of CHANS, because they are intended to enhance hydrologic services provided by biophysical systems by eliciting certain desirable behaviors from social systems through incentives that target the associated economic systems (figure 1). Although the primary objective of most PWS programs is to enhance the provision of one or more hydrologic services (e.g., water yield, flow regulation, reduction in sediment and nutrient loads; Brauman et al. 2007), such services are generated by temporally and spatially complex flows of water that interact with abiotic and biotic components, which, in turn, affect other ecosystem services, such as biodiversity, carbon sequestration and storage, biotic regulation, and recreational and aesthetic quality (Baron et al. 2002, Lele 2009). Socioeconomic systems within watersheds also exert pressures on hydrologic services that are often more complex in their spatial and temporal distribution, vary widely among different actors, and are directly influenced by the institutional structure of the programs and by the amount, distribution, and type of payments (Bosselmann and Lund 2013, Martin-Ortega et al. 2013). In addition, PWS may indirectly affect a range of conditions within the human system, including poverty levels, physical and mental health, access to diverse ecosystem services, equitability of resource distribution, and social conflict and stability (Bulte et al. 2008, Rodríguez de Francisco et al. 2013, Zheng et al. 2013). Consequently, complex feedback and nonlinear interactions often emerge within CHANS in response to PWS (figure 1) that can lead to unexpected outcomes, which, in turn, affect the degree to which stated objectives are achieved and other unintended (positive or negative) consequences occur (Srinivasan et al. 2012, Bryan 2013, Zheng et al. 2013).

More explicitly conceptualizing PWS as an integral component of the CHANS can help elucidate the most relevant and appropriate M&E indicators for assessing the progress of PWS toward achieving sustainable watershed outcomes. Our PWS–WS framework (figure 2) builds on and expands two existing analytical frameworks: the driver–pressure–state-impact–response framework (DPSIR; Carr et al. 2007, Svarstad et al. 2008) and the social–ecological systems framework (SESF; Ostrom 2007, 2009S). Both of these frameworks have features particularly relevant to enhancing water resources management. The DPSIR was designed as an interdisciplinary tool for environmental analysis, providing a structure in which physical, biological, chemical,

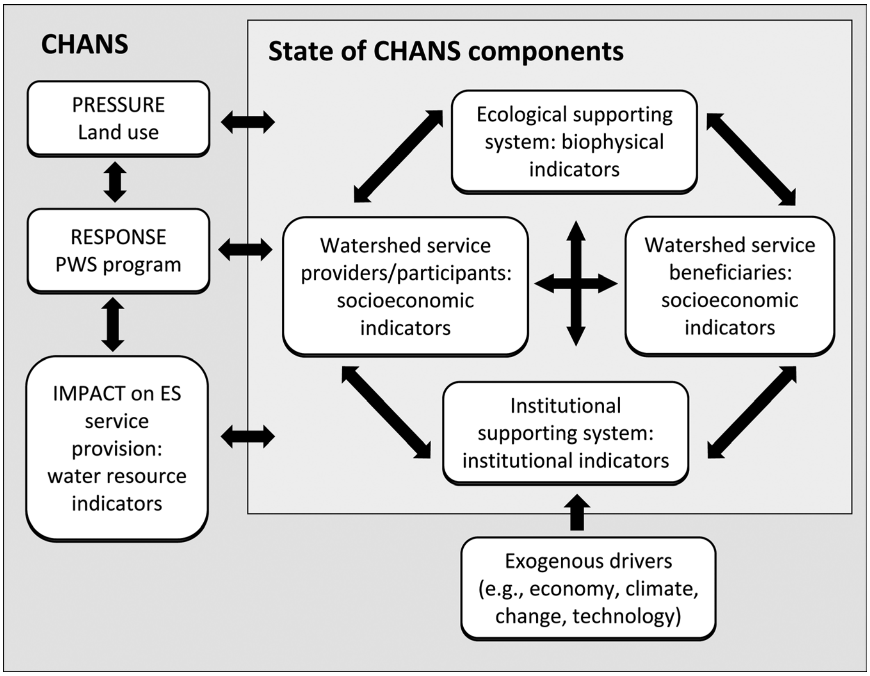

Figure 2. A conceptual framework for selecting and interpreting appropriate indicators of payments for watershed services (PWS) impacts on watershed sustainability, showing interactions between PWS policies and coupled human and natural systems (CHANS) within the watershed, across five dimensions of watershed sustainability: water resources, environment, social, economic, and institutions. Source: Adapted from the DPSIR framework as presented by Rounsevell and colleagues (2010).

and societal indicators can be analyzed and integrated to assess the achievement of policy goals, including hydrologic services (Rounsevell et al. 2010). The SESF, designed to assess progress toward the sustainable management of diverse resources (e.g., forests, fisheries, water), is based on an extensive multilevel nested hierarchy of variables associated with four major subsystems (resource systems, resource units, governance systems, and users), which, combined, are intended to explain the complex dynamics of socioecological systems (Ostrom 2009). Notably, a comparative analysis of ten widely used frameworks for assessing socioecological systems concluded that DPSIR and SESF provided the clearest guidance for the selection of indicator variables (Binder et al. 2013). Although both the DPSIR and SESF frameworks adopt an anthropocentric perspective by characterizing the ecological system as a provider of services that contributes to societal well-being, their approaches differ slightly. The SESF conceptualizes the social system by using interaction and feedback loops between both the micro and macro levels to explicitly capture the dynamics between the social and ecological systems. In contrast, the DPSIR analyzes social processes at the macro level, without explicitly addressing the more complex dynamics between the social and ecological systems, thereby using a more linear causal chain approach primarily concerned with how human action affects the ecological system. Consequently, Binder and colleagues (2013) considered DPSIR to have a greater policy-oriented focus with the goal of providing information for reducing the environmental impact of human activities, whereas the SESF is more analysis oriented, offering an organizing structure for collecting and analyzing data.

Given that PWS are both strongly action oriented and inextricably embedded within complex social–ecological systems, our PWS–WS framework (figure 2) draws heavily from both the DPSIR and SESF. We adopt the general structure of the DPSIR framework as modified by Rousenvell and colleagues (2010), intended to more clearly capture the role of humans in ecosystem service processes by incorporating a *state* variable that describes the CHANS, consisting of ecosystem service beneficiaries and providers and the environment's supporting services. The *impact* is a change in state variables, measured as a change in ecosystems-services provision and the subsequent impacts on human well-being. Our PWS–WS framework further modifies the DPSIR by explicitly considering PWS as the *response* while drawing from the SESF to incorporate a more complex social system that recognizes multiple levels of actors, interactions, and feedback, as well as a governance system—clearly an important component of watershed management and PWS outcomes. In addition, key elements unique to the design and objectives of PWS are also captured, such as the emphasis on land-use change and hydrologic-service outcomes (figure 1), while accounting for the feedback between the natural and human systems that could potentially confound or interact with PWS to affect changes in hydrologic services, the state of the CHANS, and progress toward WS.

Applying the PWS–WS framework to guiding the development of M&E to assess progress toward WS clearly reveals the importance of collecting information about indicators that capture responses from both the biophysical and human systems (figure 2). Plummer and colleagues (2012) identified five dimensions they considered representative of the broader WS literature: water resources, other physical environment, social, economic, and institutions. Considering each of these aspects in PWS M&E would bring PWS closer in alignment with the WS concept, thereby facilitating synergistic approaches for the integration of conceptual frameworks for indicator selection and application. Given the substantial overlap between the social and economic dimensions of CHANS, we elected to combine these into a single *socioeconomic* indicator group. In addition, because

PWS are policy tools aimed at eliciting certain desirable human behaviors, evaluating their performance as a policy instrument is also crucial; therefore, we added a *program performance* indicator group. To identify specific indicators for characterizing these five dimensions within our PWS–WS framework, we conducted an extensive review of M&E indicators used within the PWS and WS literatures to determine their frequency of use, to compare and contrast the degree of similarity and divergence across the PWS and WS indicators, and to identify the approaches for selecting indicators that have the greatest potential for assessing WS within a PWS context.

Comparison of WS and PWS indicators for M&E: Literature synthesis

We used a systematic approach to identify relevant papers in both the peer-reviewed and grey literatures pertaining to PWS and WS using the following key search terms: PWS: *water** or *hydrologic** service plus the modifiers *payment**, *reward**, *market**, *investment**, *compensation**; WS: *integrated water resources management**, *watershed sustainability indicators**, *watershed vulnerability indicators**, *watershed poverty indicators**. These terms were applied to the search engines Web of Science, Google Scholar, and Scopus, all commonly used in academic reviews, to identify a baseline set of papers. For the PWS literature, we also analyzed case studies listed on the Watershed Connect Project Inventory (*www.watershedconnect.com/projects*) and IIED's Watershed Markets (*http://watershedmarkets.org*). For these case studies, when possible, we identified peer-reviewed or grey literature, but more commonly, no citation was found, so we referenced the Web site where the project was described. The publications were also identified as a result of *forward* or *backward* citations associated with the above list. Only publications in English were accepted. Each body of literature was reviewed by an interdisciplinary team consisting of a social scientist and biophysical scientist, each having experience with PWS and WS. Although we recognize that our choice of databases and language may have induced geographic and ethnocultural biases, it was beyond the scope of this review to incorporate a broader distribution.

Our review yielded a total of 190 and 155 publications in the PWS and WS literatures, respectively. The following criteria were then applied as an additional filter to determine which studies met our objectives. For PWS, the criteria were the following: (a) the PWS scheme has a primary goal of sustaining watershed services (e.g., water yield, water quality, flood protection), although it may have other secondary goals as well; (b) payments are made in exchange for land-use behaviors intended to enhance water resources (e.g., forest protection, restoration) within a specific watershed; and (c) PWS have been in operation for at least one year. For WS, the criteria were the following: (a) the M&E framework or tool was focused primarily on assessing WS in relation to land-use change, because this objective is most closely aligned with the objectives of PWS; (b) guidelines were included for identifying and selecting relevant indicators of WS; and (c) the WS indicators were applied to empirical case studies at the watershed scale. As several studies were often conducted of the same PWS scheme, to evaluate the full range of indicators, we organized the PWS literature by scheme rather than individual study. This was not necessary for the WS literature, because none of the selected publications considered the same geographic location.

This filtering process yielded a total of 62 PWS schemes and 57 WS publications for the final synthesis (see supplemental S1 for details). Each accepted WS publication or set of PWS publications was scored according to measured indicators. This process yielded a master list of 55 indicators, classified according to the natural and human dimensions of the PWS–WS framework, which were then merged into a final list of 5 indicator groups and 21 specific indicators (tables 1a, b, c) to facilitate the comparative analysis.

Comparative analysis and emergent patterns of indicators within the PWS and WS literatures

Overall, *water resources* and *other physical/environmental* indicators were more commonly cited than *socioeconomic*, *program performance*, and *governance systems* indicators for both PWS and WS literatures (figure 3). A large proportion (more than 60%) of publications in both literatures measured some aspect of water resources; the majority of both PWS and WS included surface water yield and water quality indicators, with PWS having slightly more surface water yield indicators (66% versus 60%) and substantially more water quality indicators (79% versus 54%). This trend may reflect a tendency of PWS programs to respond to actual or anticipated water quality degradation, because forest conversion to other land uses is generally associated with negative impacts on water quality (Aylward 2005). In contrast, the relationship between forest cover and water quantity benefits is less clear, with the possible exception of cloud forest (Bruijnzeel et al. 2005). Very few PWS schemes measured stream flow variability, whereas almost half of the WS papers did so. This finding may relate to the longer duration of most WS studies compared with PWS schemes and therefore the greater availability of intra- and interannual stream discharge variability data (e.g., 12 out of 57 WS papers had at least 32 years of data, whereas about half of the PWS studies were less than 10 years old and 20% were less than 5 years old). The lack of available financial resources for PWS monitoring may also have precluded the purchase of continuous stream gauges. Finally, neither PWS nor WS included many groundwater supply indicators, although for WS, the frequency (21%) was about five-fold greater than it was for PWS. This result may reflect that many PWS schemes are located in watersheds where surface water is relatively plentiful and groundwater supplies are less relevant, as well as a lack of readily available groundwater data in many less industrialized countries, where PWS programs have been predominantly implemented (Porras et al. 2008).

Table 1a. Natural system dimension: Definitions of biophysical indicators.

Indicator group	Indicators	Definition of indicator	Examples of measurable variables
Water resources	Surface water supply	Average surface water availability at annual or interannual time scales	Supply relative to demand or as absolute quantities
	Quality	Water quality constituents related to human or aquatic ecosystem health	Absolute quantities of constituents, constituent quantities relative to a water quality target, or spatial quantities of surface water bodies not in compliance with water quality target
	Stream flow variability	Intraannual or interannual streamflows	Baseflows (e.g., flow corresponding to 10% recurrence interval or average over low flow month), peak flows (e.g., flow corresponding to 95% recurrence interval)
	Groundwater supply	Aquifer recharge	Aquifer recharge relative to groundwater withdrawals or groundwater availability
Other physical/ environmental	Land use	Fraction of total land area currently associated with land uses	Land area or percentage of the landscape that is either converted, restored or maintained in a particular land use (e,g, forested, agricultural, or urban categories)
	Other ecosystem services	Ecosystem services not directly related to hydrologic services	Carbon storage (total biomass or sequestration rate per area) or biodiversity (e.g., total number of species, ratio of native to exotic species, number of rare or threatened species, expressed on an area basis)
	Soil management	Measures related to the protection of physical and biological aspects of soil	Soil erosion, soil productivity, fraction of land where best management practices are applied, soil hydraulic conductivity, soil infiltration rates, soil salinity, and soil subsidence

Table 1b. Human system dimension: Definitions of socioeconomic indicators.

Indicator group	Indicators	Definition of indicator	Examples of measurable variables
Socioeconomic	Knowledge and participation	Awareness and knowledge about the program; perceptions and motivations for participation in program; environmental knowledge	Number of households aware of or participating in the program; percentage of households reporting favorable perceptions of program
	Water demand and access	Need for and access to water and sanitation	Water access per capita; number of households with piped water
	Land management practices	Type of land management practices and their consequences for water resources and social dynamics	Ratio of irrigated land to total land, traditional versus modern land management practices
	Human health	Human health and disease incidence	Prevalence of water related disease
	Demographics	General characteristics of upstream and downstream communities	Population density; gender; years of schooling
	Livelihoods	Household production consumption activities	Number of households engaged in agriculture
	Social conflict	Focus on social relations, conflict, power	Water use and management conflicts
	Poverty	Income and poverty alleviation measures	GDP per capita; household durables, assets, consumption, expenditures
	Labor	Jobs and labor	Percentage of population employed in off-farm jobs; number of household members involved in farm work
	Equity and fairness	Concern about the distribution of program costs and benefits	Number of poor participants in program

For the *other physical/environmental* indicator group, almost 100% of PWS schemes and two-thirds of WS studies included a land-use measure (figure 3). The frequency of this indicator is not surprising for PWS, because most schemes are tied to land use–based payments and land-cover data are commonly available. Roughly the same—relatively small—fraction of PWS and WS included measures of other ecosystem services, most commonly, carbon storage (two WS and seven PWS) and biodiversity (seven WS and nine PWS), thereby limiting their ability to fully assess benefit–cost trade-offs across multiple services. WS tended to use indicators of soil management more than PWS schemes did (35% versus 22%, respectively), possibly because of a combination of limited resources for PWS M&E and prioritization of direct measures of hydrologic services.

For the human dimension, more than 30% of WS measured water demand and access and demographic variables, compared with 5% of PWS (figure 3). WS was also more likely than PWS to include indicators related to labor (6% versus 3%, respectively). These indicators, which help gauge

Table 1c. Human system dimension: Definitions of program performance and governance indicators.			
Indicator group	**Indicators**	**Definition of indicator**	**Examples of measurable variables**
Program performance	Program expenditures	Total costs spent on implementing, monitoring and evaluating the program	Amount of funding and/or other resources allocated to the program
	Economic efficacy	Economic costs and benefits of the program	Opportunity costs of participants, willingness to pay by beneficiaries, additionality of program on land-use change
Governance system	Land tenure/ property rights systems	Existence of formal or informal property rights	Percentage of population with formal land title
	Social organizational capacity	Ability of government, NGOs, or other administrating entity to implement programs and ensure compliance; role played by intermediaries in water transactions	Number of government officials; past performance; number of nongovernmental organizations; level of social capital; presence and enforcement of legal structures

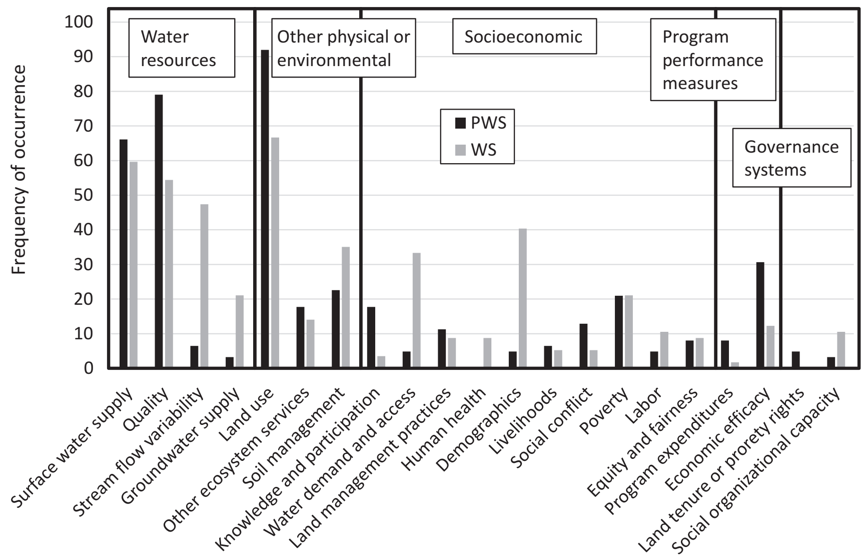

Figure 3. The frequency of occurrence (as a percentage) of indicators across payment for watershed services (PWS) schemes and watershed services (WS) papers, organized by dimension.

watershed vulnerability to changes in human system dynamics, are typically not difficult to measure but may not be included in PWS because they are not directly related to program outcomes. The one exception was PWS papers on China's Sloping Lands Program, which has a specific objective of increasing off-farm labor (Zhen et al. 2014). Surprisingly, no PWS schemes measured human health outcomes, compared with 9% of WS, despite the potential importance of the health benefits resulting from positive impacts of PWS on water quality. Land-management and livelihood measures were collected with similar frequency across PWS and WS. Poverty, a prominent sustainability issue within both PWS and WS (e.g., Pagiola et al. 2005), was the most frequent variable assessed for PWS (21%) and similar for WS. PWS more frequently included measures of knowledge and participation than WS did (18% versus 4%, respectively). Finally, PWS was more likely to measure social conflict than did WS (13% versus 5%, respectively); this measure typically relates to conflicts resulting from underlying power relations and is perhaps more applicable to PWS given the focus on motivating behavioral change through financial incentives. The relatively low frequency of equity measurements (less than 10%) was surprising given its emphasis within PWS (Engel et al. 2008, Pascual et al. 2010). Although we initially anticipated that WS would be more inclusive of socioeconomic indicators than PWS would be, the difference was not particularly great, with the exception of water demand and access, demographics, and human health indicators.

As anticipated, we found that PWS schemes were more likely than the WS literature to include indicators related to the *program performance*, because PWS schemes are a specific policy response, whereas the WS papers are typically focused more generally on watershed management. The frequency of economic-efficacy measures within the PWS literature was relatively high (31%), which was not surprising given the importance of the conditionality and additionality of outcomes in PWS schemes (Engel et al. 2008). Program expenditures or equity measures were rarely collected (less than10%) in either the PWS or WS literatures. We suspect that information on program expenditures is available but not reported in the published literature; a more transparent understanding of the costs of these schemes would be valuable in assessing benefitcost relationships and in designing new or improving existing PWS programs.

Overall, *governance systems* was the most poorly represented indicator group (figure 3), notwithstanding the emphasis of the importance of governance- or institutional-related factors in both the PWS and WS literatures. Challenges in defining and measuring governance-related variables may have contributed to their relatively low frequency in WS and PWS M&E efforts (Kjaer 2004). For instance, there is an increasing recognition that intermediaries, such as nongovernmental organizations, are often crucial to the success of PWS schemes; however, obtaining reliable measurements of the contributions of intermediaries to PWS program success in not trivial (Schomers et al. 2015). Direct measures of property rights were more common for PWS, which is likely related to the importance of clear land tenure arrangements before payments are made (Naughton-Treves and Wendland 2014), an aspect that is less relevant for WS.

When we compared measurements across dimensions and indicator groups, the most notable observation was that the overall frequency of *water resources* or *other physical/environmental* indicators is much greater than the *socioeconomic*, *governance*, and *program performance* indicators (figure 3). The magnitude of this difference is striking, with no single indicator on the social science side having a frequency of greater than 40%, whereas three indicator groups—water yield, quality, and land use—were measured with more than 50% frequency. This result could be due to an inherent bias, given the programmatic emphasis on achieving water-resources outcomes; certainly for PWS, there may be a more pressing need to direct funds at monitoring the target output—hydrologic services—that are being paid for by downstream users. Overall, these findings suggest a potential limitation of both the PWS and WS literatures to account for the broader aspects of WS that require an understanding of the state and dynamics of the CHANS (i.e., figure 2). Another factor may be the greater range of indicators included in the human dimension (i.e., eight indicator groups versus three indicator groups in the natural dimension), such that a given number of indicators may simply have been spread across more groups. The different approaches to defining and categorizing indicator groups can make the interpretation and integration of natural and human dimensions in M&E frameworks challenging and lead to trade-offs between the breadth and depth of information obtained (e.g., Plummer et al. 2012). However, this explanation would not hold for measures of the governance system, which still experienced a low frequency of measurement despite having only two indicator groups.

Case study analysis: Applying the PWS–WS framework to monitoring and evaluation

The above comparative analysis of the PWS and WS literatures revealed that despite a few differences in the frequency of the use of certain indicators, the patterns were fairly similar. Moreover, although including indicators from the human systems dimensions within M&E efforts provided valuable additional information about progress toward WS not revealed by only using indicators from the biophysical dimensions, our findings also underscored the importance of selecting indicators that are place based, context specific, and locally meaningful. In this section, we apply the PWS–WS framework (figure 2) to two case studies to illustrate how a contextualized analysis of the framework components can guide the selection of appropriate M&E indicators for understanding the complex dynamics between PWS and CHANS and their consequences for WS.

We first consider China's Paddy Land-to-Dry-Land (PLDL) PWS program described by Zheng and colleagues (2013), which compensates upstream rural communities for providing hydrologic services (water quality and quantity) to downstream urban consumers. If our PWS–WS framework is applied, large-scale rice cultivation by upstream farmers near the Miyun Reservoir (the only surface water source for domestic water in Beijing) is *pressure* on the CHANS, leading to declining water yield and nitrogen (N) and phosphorous (P) contamination adversely affecting downstream watershed service beneficiaries. The *response* by the Chinese government was to create the PLDL program, which seeks to incentivize upstream farmers through direct payments to adopt land-use practices more favorable to water resources, particularly dryland crops such as corn. The direct impacts on target water resources were assessed by estimating the changes in water yield and water quality (table 2), using data available from previous studies. Water yield was estimated as the difference between rainfall and evapotranspiration, using potential evapotranspiration values for rice and corn derived from models, and assuming relatively small catchment water losses from deep percolation and leakage. Nutrient export in runoff (i.e., N and P concentrations and loads) was measured empirically for a different watershed in China having similar land-use characteristics, with adjustments for fertilizer use obtained from household surveys (see below). Findings suggested that, as a result of land-use conversion from rice paddies to corn promoted by the PLDL program, water yield increased by 5%, and total N and P loads decreased by 0.9% and 2.6% reduction, respectively.

Several indicators of the indirect impacts of the PDLD program on socioeconomic dimensions were also assessed (table 2) using household surveys to evaluate changes in livelihood activities and the associated cost–benefit tradeoffs for the natural and human systems. Opportunity cost, calculated as the difference between gross income and the cost of planting rice versus corn, showed that PWS received by landowners were about 1.2 times greater than the opportunity cost to convert from rice cultivation to corn. The benefits from estimated water quantity and quality improvements to downstream Beijing consumers were much higher than the total program costs, resulting in an estimated overall benefit–cost ratio of 1.5. Although agricultural income to farmers who switched from rice to corn decreased by 2000 yuan per year (y), this was more than offset by earnings of 3000 yuan per y from migrant activities, which increased because of the reduced labor requirements of dryland crops. Although improved household income led to increased education spending by PLDL participants—a potential positive impact on future livelihoods—expenditures on household goods and fossil fuel consumption also increased, with possible negative environmental effects related to greenhouse gas emissions. Fertilizer and pesticide application also increased under corn cultivation,

Table 2. Conceptual framework for selecting and interpreting appropriate indicators of payments for watershed services (PWS) impacts on watershed sustainability applied to China[a] and Ecuador[b] case studies.

	Indicators	
Drivers	**China case study**	**Ecuador case study**
Pressure	• High water and fertilizer use by upland rice cultivation results in reduced water yield and quality	• Clearing and conversion of forests to crops or pasture by upstream farmers threatens downstream water supply
Response	• PWS PLDL—payments to upstream landowners to convert from rice to dryland crops	• PWS program to pay upstream farmers to conserve forests and restore degraded lands (e.g., agroforestry, natural regeneration).
Impact—water resources indicators	• Changes in water yield and water quality	• Changes in water yield and water quality
State: Ecological supporting system—other physical or environmental indicators	• Greenhouse gas emissions • Ground water contamination	• Impacts of land-use practices on soil degradation • Leakage (clearing of forest outside area)
State: Watershed service providers—socioeconomic indicators	• Spending on education, material goods, energy (wood vs fossil fuels) • Change in land-use practices from rice to corn cultivation • Livelihood activities/labor • Opportunity cost (rice versus corn cultivation) • Income from different sources (migrant labor) • Willingness to continue with new practices if payments stop	• Opportunity cost (farming versus conservation) • Equity in access to water resources • Equity in PWS payments • Changes in social/cultural traditions (e.g., migration, traditional land-use system fallows) • Social conflict; power structures
State: Watershed service beneficiaries—socioeconomic indicators	• Improvements in water quality and supply • Cost of the PLDL program	• Water supply • Payments to support the PWS program
State: Institutional supporting system—governance system indicators		• Community organization and capacity • Legal system for water rights and land use; enforcement

Abbreviations: PLDL, China's Paddy Land-to-Dry-Land PWS program; PWS, payment for watershed services. [a]Zheng et al. (2013). [b]Echavarria et al. (2004), Wunder and Alban (2008), Quintero et al. (2009), Rodríguez de Francisco et al. (2013).

but because of the higher nutrient export coefficients for rice compared with those for corn, there was an estimated net positive effect on nutrient export. Although the PLDL program has apparently increased household well-being according to many of the criteria assessed, the analysis also revealed potential unintended indirect consequences and trade-offs. For example, two impacts considered positive—improved income from increased migrant labor and reduced nutrient export under corn cultivation—may also have associated unintended (and unquantified) negative consequences, because increased household wealth led to greater fossil fuel and fertilizer consumption, which may have detrimental impacts on climate change, environmental quality, and human health.

Overall, the study by Zheng and colleagues (2013) highlighted the value of incorporating both biophysical and socioeconomic indicators to effectively capture the key relationships between PWS and CHANS (figure 2) that may affect long-term program success and sustainability in reaching WS outcomes. Although greater livelihood diversification and mobility amongst PWS program participants may have led to greater socioeconomic benefits (e.g., greater resilience to and ability to cope with changing markets or climate conditions), these benefits may be counteracted in the long term by other negative impacts that reduce WS and that would be useful to include in future evaluations, such as the environmental impacts of increased fertilizer use or the implications of increased migrational labor on social organization or cultural traditions. In addition, a high percentage (88%) of program participants said they would revert back to rice farming if payments stopped, suggesting the limited long-term impacts on participants' behaviors related to sustainable land-use change. Other socioeconomic indicators from the PWS–WS framework not included in this study, such as measures of social conflict, equity, knowledge, and governance, may have provided insight about the motivations underlying program participation. Consequently, using the PWS–WS framework early on in the M&E planning and implementation stages can facilitate the identification of the most important indicators for elucidating key interactions and trade-offs between biophysical and socioeconomic dimensions within CHANS relevant to WS outcomes.

Applying the PWS–WS framework to a second case study of PWS in Ecuador illustrates the potential consequences of relying on a more narrowly defined set of indicators when monitoring and evaluating PWS program performance. In this case, the perceived risks to water quality and quantity by the downstream municipality of Pimampiro attributed to the conversion of upstream forests to agriculture by the upstream community of Nueva America were creating *pressures* to change upstream land-use behaviors, eventually leading to a *response* by the Ecuadorian government in 1999 to create a PWS scheme (table 2; Echavarria et al. 2004, Wunder and Alban 2008, Quintero et al. 2009, Rodríguez de Francisco et al. 2013). Several studies assessed the Pimampiro PWS program using different sets of indicators. Wunder and Alban (2008) and Quintero and colleagues (2009) used hydrologic and economic measures to examine (a) the relationship between the incremental area conserved or restored and marginal ecosystem service gains and (b) how PWS payments compared with farmers' estimated opportunity costs. The analysis was based on a modeling approach that simulated water yield and sediment loss under different land-use scenarios (e.g., forest conservation versus conversion to annual crops and pastures). Model calibration was conducted by adjusting the model parameters to achieve the best possible correspondence between observed and simulated streamflow data at the basin outlet. The results suggested that in the absence of the existing PWS scheme, resumed deforestation would increase sediment loads by more than 50%. Although forest conservation reduced dry-season water yield because of lower evapotranspiration by alternative land-use systems, the overall reduction was small (0.5%), largely because forests maintained higher infiltration, lateral flow, and groundwater recharge (Quintero et al. 2009). In addition, an economic analysis using a linear programming model to optimize net income from different land-use systems suggested that continued deforestation would provide a slightly higher farming income than receiving PWS to conserve forests, and therefore, current PWS payments were below farmers' opportunity costs. Several factors were identified that may have contributed to farmers opting to enroll in PWS programs, including a preference for a stable, risk-free income and anticipated stricter enforcement of forest protection laws. Combined, these findings were interpreted as indicating a high degree of PHS program efficiency and net benefits due to impacts on avoided sedimentation and enhanced water quality (considered the most important hydrologic service) and the relatively small difference between PHS payments and opportunity costs.

In contrast, Echavarria and colleagues' (2004) assessment of the Pimampiro PWS program focused more strongly on socioeconomic and program performance indicators. The results revealed several potential indirect negative consequences for program participants, including dissatisfaction with payment levels and diminished local social-organizational capacity, countered by an overall satisfaction by downstream urban water consumers with the current water services obtained in exchange for the PWS water tax. A more recent analysis by Rodríguez de Francisco and colleagues (2013), aimed at understanding the dynamics between the PWS program and the local community, used semistructured interviews and focus group discussions with peasant farmers in Nueva America to assess a broader range of the indicators of the socioeconomic and governance dimensions, including equity, fairness, poverty, legal structures, level of social organization and social capital, and social conflict. The results suggested that poor forest landowners viewed the PWS program with great suspicion and were overall dissatisfied with its implementation. For example, a legal system that established water rights for downstream community members that had been in place prior to the settlement of Nueva America greatly restricted access by upstream farmers to water resources. The PWS scheme apparently further reinforced this situation by institutionalizing existing inequalities over access to water and social power structures. Consequently, landowners with large parcels in highly remote forests that were not accessible or profitable for farming benefited greatly from PWS given the higher payments allocated to forest lands, whereas farmers with small forest parcels or degraded lands closer to the community and with higher agricultural value (i.e., opportunity costs) benefited less from the lower payments provided by the PWS scheme for these lands. Moreover, average payments were set far below the average opportunity cost on the basis of a perceived unfair negotiation process. This situation was further exacerbated by the perception among landowners that participation in the PWS scheme was not voluntary (as initially promoted), as many felt coerced to join because of possible legal enforcement of a forestry law that prohibited the clearing of forestlands. The results suggested that rather than reducing poverty and inequality within the poor and marginalized communities targeted by the PWS program, the PWS program reinforced existing inequalities, power relationships, and social conflict between upstream and downstream communities. In addition, the intensification of land use within smaller areas and the leakage of land-use conversion to surrounding areas increased the degradation of the biophysical system.

The sharply contrasting conclusions drawn from the above evaluations of the Ecuadorian PHS case study demonstrate how integrating information about both biophysical and socioeconomic indicators provides a deeper and more contextualized understanding of the PHS impacts on CHANS dynamics, compared with using information from only one of the two dimensions. In this case, the unintended, indirect impacts of PWS on the socioeconomic well-being of upstream communities and environmental quality threatened to undermine the long-term achievement of WS goals (Echavarria et al. 2004, Rodríguez de Francisco et al. 2013). This is in contrast to the positive assessments of the program obtained from a less holistic analysis of direct PWS impacts on land cover, hydrologic services, and economic efficiency (Wunder and Alban 2008, Quintero et al. 2009).

An important consideration revealed through the comparative analysis of the Chinese and Ecuadorian case studies is that the selection of appropriate indicators under any of the five dimensions in the PWS–WS framework depends heavily on having sound baseline knowledge about the local conditions and program objectives related to the pressures on the CHANS. In other words, PWS–CHANS interactions are fundamentally contextual and site specific, requiring an in-depth understanding of local societal dynamics and relationships to environmental conditions and services. Examples include the indicators related to expenditures on material goods or income from migrant labor—which were highly relevant in the China example, but did not apply to the Ecuador example, and the perceived inequalities and unfairness associated with underlying power relationships amongst Ecuadorian farmers that threatened the long-term success of the Pimampiro PHS scheme—apparently of less importance in the China PHS program. Also noteworthy is that there could potentially be numerous indicators under any one dimension, and inevitably some important indicators may be overlooked. For instance, although more holistic in coverage across the dimensions, the China case study did not assess many of the socioeconomic indicators of power, social conflict, equity, and knowledge that were assessed in the Ecuador case study, and their inclusion in future assessments may provide additional insights about CHANS dynamics and WS in response to PWS.

Detecting change in water cv, water quality and other ecosystem services as a result of the implementation of PWS has not been well documented in the literature, especially using quantitative measures (Brouwer et al. 2011). We found no studies that documented a change in streamflow or water quality based on field measurements collected from watersheds participating in PWS programs. In our two case studies, attempts were made to ascertain improvements in streamflow yield and water quality, but both studies relied on secondary data sources, potentially introducing uncertainties affecting the quality, relevance, and credibility of the data. In the China example, runoff data were obtained from research conducted in a large watershed located outside the study area (Zhang et al. 2013), which may have introduced errors due to varying biophysical, climatic, and management conditions. Detection limits posed by the high degree of variability inherent in streamflow measurements may have introduced additional uncertainties. For instance, using stage–discharge measurements to quantify streamflow can lead to errors of 6–19% (Harmel et al. 2006), which is above the 5% increase in yield estimated by Zhang and colleagues (2013). Nutrient loads are suspect to uncertainty in streamflow, sample collection, sample preservation and storage, and sample analysis. Harmel and colleagues (2006) estimated that uncertainty in TN loads ranges from 11–70% and TP loads from 8–110%; again, these possible errors are greater than the reported reduction of 0.9% and 2.6% in total N and P loads reported in Zhang and colleagues (2013), indicating that detecting differences as a result of the PLDL would have been unlikely. In the Ecuador example, a modeling approach was used to estimate the improvements in ecosystem services on the basis of simulated changes in land use (Quintero et al. 2009). Although models can offer insights into the impacts of PHS policy schemes and subsequent land-use change, existing field observations are often not sufficient to generate robustly calibrated models. In the Ecuadorian case study, model calibration was based on a relatively small streamflow observation data set, poor agreement between modeled and observed streamflows (Nash-Sutcliffe coefficient of 0.03), and no sediment loss observations. The danger, of course, is that poorly calibrated models may under- or overestimate impacts of PHS programs on significant indicators such as peak flows, dry-season flows, sediment loss, and other water quality constituents.

The above analysis underscores the importance of field data collection to reliably measure changes in hydrologic services in response to PHS. Ideally, such efforts should employ a paired watershed or a before–after approach, which involve monitoring watershed conditions for some period of time before implementing the PWS program. In the former, typically 5–10 years of data are required prior to treating one of the paired watersheds after relationships have been developed between the two watersheds, whereas for the latter, the calibration period is longer because of the effect natural variability has on uncertainty and the subsequent detection of differences (Loftis et al. 2001). In summary, given the high degree of uncertainty in hydrologic measurements and the added complexities of the geographies where PWS are implemented, more sophisticated M&E programs are needed to improve the detection of PWS impacts on hydrologic services, which will require overcoming existing constraints related to the availability of funding and technical expertise.

Conclusions

Lack of an integrated, holistic approach to understanding how PWS influence WS—taking into account both the dynamic interactions within CHANS in addition to the target ecosystem service—is contributing to the growing uncertainty over whether PWS provide an effective mechanism for improving the quality and supply of water resources without detrimentally affecting the CHANS, thereby promoting the long-term health and sustainability of a watershed's natural and human systems. In this article, we developed a new framework for PWS that explicitly considers PWS as part of the CHANS. With this framework we identified five broad dimensions—water resources, other physical/environmental, socioeconomic, program performance, and governance system—that guide the selection of indicators for assessing PWS impacts on WS. We explored the conceptual congruence between the WS concept, which has been an integrating concept in watershed management for decades, and PWS by highlighting the types of indicators commonly used to assess outcomes in both of these literatures. Despite a pattern of water resource indicators, particularly water quality and water quantity, being more frequently used in the PWS literature, whereas socioeconomic indicators associated with water service providers and beneficiaries were assessed more frequently within the WS literature, the differences were not as great as expected. We applied the PWS–WS framework to two PWS case studies to reveal the need to include relevant biophysical and human system indicators to better understand the underlying feedback and interactions within CHANS and the resulting trade-offs between desirable and unintended consequences for WS. Application of the PWS–WS framework also reinforced the importance of a place-based, contextualized approach for selecting the most meaningful indicators to best capture the dynamics of the CHANS in response to PWS. Finally, we discussed the importance of including actual long-term measures of streamflow yield and water quality to establish baseline conditions, detect response to PWS intervention, and ensure data quality. As interest in incorporating M&E as an integral component of PWS programs increases, the PWS–WS framework provides a useful tool for guiding interdisciplinary efforts to enhance understanding of the complex drivers, interactions, and feedback that determine the potential for CHANS to achieve long-term goals related to hydrologic services while positively influencing the well-being of upstream and downstream communities.

Acknowledgments
The authors would like to thank Sophie Avila-Foucat, Russ Congalton, and Robert Manson for contributing to early discussions about PWS-CHANS in Mexico that inspired our thinking for this paper, Mariana Zareth Nava Lopez for assembling and summarizing the literature, and Z. Carter Berry for reviewing and providing valuable feedback on an earlier draft of the manuscript. The authors would like to acknowledge NSF's Dynamics of Coupled Natural-Human Systems (CNH) program grant no. 1313804 for partial funding in support of the research conducted for this synthesis.

Supplemental material
The supplemental material is available online at *http://bioscience.oxfordjournals.org/lookup/suppl/doi:10.1093/biosci/biv051/-/DC1*.

References cited

Aylward B. 2005. Land use, hydrological function, and economic valuation. Pages 99–120 in Bonell M, Bruijnzeel LA, eds. Forest Water and People in the Humid Tropics: Past, Present, and Future Hydrological Research for Integrated Land and Water Management. Cambridge University Press.

Baron JS, Popp NL, Angermeier PL, Dahm CN, Gleick PH, Hariston NG Jr, Jackson RB, Johnston CA, Richter BD, Steinman AD. 2002. Meeting ecological and societal needs for freshwater. Ecological Applications 12: 1247–1260.

Binder CR, Hinkel J, Bots PWG, Pahl-Wostl C. 2013. Comparison of frameworks for analyzing social–ecological systems. Ecology and Society. 18 (art. 26). (8 April 2015; *http://dx.doi.org/10.5751/ES-05551-180426*)

Brauman KA, Daily GC, Duarte TK, Mooney HA. 2007. The nature and value of ecosystem services: An overview highlighting hydrologic services. Annual Review of Environmental Resources 32: 67–98.

Brouwer R, Tesfaye A, Pauw P. 2011. Meta-analysis of institutional–economic factors explaining the environmental performance of payments for watershed services. Environmental Conservation 38: 380–392.

Bruijnzeel LA, Bonell M, Gilmour DA, Lamb D. 2005. Conclusion: Forests, Water and People in the Humid Tropics: An emerging view. Pages 906–925 in Bonell M, Bruijnzeel LA, eds. Forest Water and People in the Humid Tropics: Past, Present, and Future Hydrological Research for Integrated Land and Water Management. Cambridge University Press.

Bryan BA. 2013. Incentives, land use, and ecosystem services: Synthesizing complex linkages. Environmental Science and Policy 27: 124–134.

Bulte E, Lipper L, Stringer R, Zilberman D. 2008. Payments for ecosystem services and poverty reduction: Concepts, issues, and empirical perspectives. Environment and Development Economics 13: 245–254.

Chaves HML, Alipaz S. 2007. An integrated indicator based on basin hydrology, environment, life, and policy: The watershed sustainability index. Water Resources Management 21: 883–895.

Chorley RJ, Beckinsale RP. 1971. Introduction to Geographical Hydrology. Methuen.

Dale VH, Beyeler SC. 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1: 3–10.

Daw T, Brown K, Rosendo S, Pomeroy R. 2011. Applying the ecosystem services concept to poverty alleviation: The need to disaggregate human well-being. Environmental Conservation 38: 370–379.

Echavarria M, Vogel J, Albán M, Meneses F. 2004. The Impacts of Payments for Watershed Services in Ecuador: Emerging Lessons from Pimampiro and Cuenca. International Institute for Environment and Development.

Engel S, Pagiola S, Wunder S. 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65: 663–674.

Gleick PH. 1998. Water in crisis: Paths to sustainable water use. Ecological Applications 8: 571–579.

Gober P, Wheater HS. 2014. Sociohydrology and the science–policy interface: A case study of the Saskatchewan River Basin. Hydrology and Earth System Sciences 18: 1413–1422.

Goldman-Benner RL, Benitez S, Boucher T, Calvache A, Daily G, Kareiva P, Kroeger T, Ramos A. 2012. Water funds and payments for ecosystem services: Practice learns from theory and theory can learn from practice. Oryx 46: 55–63.

Harmel RD, Cooper RJ, Slade RM, Haney RL, Arnold JG. 2006. Cumulative uncertainty in measured streamflow and water quality data for small watersheds. Transactions of the ABASE 49: 689–701.

Heinz I, Pulido-Velazquez M, Lund JR, Andreu J. 2007. Hydroeconomic modeling in river basin management: Implications and applications for the European Water Framework Directive. Water Resources Management 21: 1103–1125.

Jack BK, Kousky C, Sims KRE. 2008. Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. Proceedings of the National Academy of Sciences 105: 9465–9470.

Jimenez-Cisneros B. 1996. Water availability index based on quality and quantity: Its application in Mexico. Water Science and Technology 34: 165–172.

Juwana I, Perera BJC, Muttil N. 2010. A water sustainability index for West Java. Part 1: Developing the conceptual framework. Water Science and Technology 62: 1629–1640.

Kang M-G, Lee G-M. 2011. Multicriteria evaluation of water resources sustainability in the context of watershed management. Journal of the American Water Resources Association 47: 813–827.

Karr JR. 1991. Biological integrity: A long neglected aspect of water resource management. Ecological Applications 1: 66–84.

Kenney DS. 1999. Historical and sociopolitical context of the western watersheds movement. Journal of the American Water Resources Association 35: 493–503.

Kjaer AM. 2004. Governance. Polity.

Kneese AV. 1964. The Economics of Regional Water Quality Management. Johns Hopkins.

Kosoy N, Martinez-Tuna M, Muradian R, Martinez-Alier J. 2007. Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. Ecological Economics 61: 446–455.

Lele S. 2009. Watershed services of tropical forests: From hydrology to economic valuation to integrated analysis. Current Opinion in Environmental Sustainability 1: 148–155.

Liu J, et al. 2007. Coupled human and natural systems. Ambio 36: 639–649.

Locatelli B, Vignola R. 2009. Managing watershed services of tropical forests and plantations: Can meta-analysis help? Forest Ecology and Management 258: 1864–1870.

Loftis JC, MacDonald LH, Streett S, Iyer HK, Bunte K. 2001. Detecting cumulative watershed effects: The statistical power of pairing. Journal of Hydrology 251: 49–64.

Martin A, Gross-Camp N, Kebede B, McGurie S, Munyarukaza J. 2013. Whose environmental justice? Exploring local and global perspectives in a payments for ecosystem services scheme in Rwanda. Geoforum 54: 167–177.

Martin-Ortega J, Ojea E, Roux C. 2013. Payments for water ecosystem services in Latin America: A literature review and conceptual model. Ecosystem Services 6: 122–32.

Naughton-Treves L, Wendland K. 2014. Land tenure and tropical forest carbon management. World Development 55: 1–6.

Neary DG. 2000. Changing perceptions of watershed management from a retrospective viewpoint. Pages 167–176 in Ffolliott PF, Baker MB Jr, Edminster CB, Dillon MC, Mora KL. Land Stewardship in the 21st Century: The Contributions of Watershed Management. US Department of Agriculture Forest Service. Proceedings no. RMRS-P-13.

Ostrom E. 2007. A diagnostic approach for going beyond pancreas. Proceedings of the National Academy of Sciences 104: 15181–15187.

———. 2009. A general framework for analyzing sustainability of social–ecological systems. Science 325: 419–22.

Pagiola S. 2008. Payments for environmental services in Costa Rica. Environmental Economics 65: 712–724

Pagiola S, Arcenas A, Platais G. 2005. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development 33: 237–253.

Pascual U, Muradian R, Rodríguez LC, Duraiappah A. 2010. Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics 69: 1237–1244.

Plummer R, De Loe R, Armitage D. 2012. A systematic review of water vulnerability assessment tools. Water Resources Management 26: 4327–4346.

Porras I, Aylward B, Dengel J. 2013. Monitoring Payments for Watershed Services Schemes in Developing Countries. International Institute for Environment and Development. MPRA Paper no. 47185. (8 April 2015; *http://mpra.ub.uni-muenchen.de/47185*)

Porras I, Grieg-Gran M, Neves N. 2008. All that glitters: A review of payments for watershed services in developing countries. Natural Resource Issues No.11. International Institute for Environment and Development, London, UK.

Quintero M, Wunder S, Estrada RD. 2009. For services rendered? Modeling hydrology and livelihoods in Andean payments for environmental services schemes. Forest Ecology and Management 258: 1871–1880.

Reyers B, Biggs R, Cummings GS, Elmqvist T, Heinowicz AP, Polasky S. 2013. Getting the measure of ecological services: A social–ecological approach. Frontiers in Ecology and Environment 11: 268–273.

Rodríguez de Francisco JC, Budds J, Boelens R. 2013. Payment for environmental services and unequal resource control in Pimampiro, Ecuador. Society and Natural Resources 26: 1217–1233.

Rounsevell MDA, Dawson TP, Harrison PA. 2010. A conceptual framework to assess the effects of environmental change on ecosystem services. Biodiversity and Conservation 19: 2823–2842.

Schomers S, Sattler C, Matzdof B. 2015. An analytical framework for assessing the potential of intermediaries to improve the performance of payments for ecosystem services. Land Use Policy 42: 58–70.

Shapiro-Garza E. 2013. Contesting the market-based nature of Mexico's national payments for ecosystem services programs: Four sites of articulation and hybridization. Geoforum 46: 5–15.

Sivapalan M, Savenije HHG, Bloschl G. 2012. Sociohydrology: A new science of people and water. Hydrological Processes 26: 1270–1276.

Sood A, Ritter WF. 2011. Developing a framework to measure watershed sustainability by using hydrological/water quality model. Journal of Water Resource and Protection 3: 788–804.

Srinivasan V, Lambin EF, Gorelick SM, Thompson BH, Rozelle S. 2012. The nature and causes of the global water crisis: Syndromes from a meta-analysis of coupled human–water studies. Water Resources Research 48 (art. W10516). doi:10.1029/2011WR011087

Su C, Fu B, Wei Y, Lu Y, Liu G, Wang D, Mao K, Feng X. 2012. Ecosystem management based on ecosystem services and human activities: A case study in the Yanhe watershed. Sustainability Science 7: 17–32.

Sullivan CA. 2011. Quantifying water vulnerability: A multidimensional approach. Stochastic Environmental Research Risk Assessment 25: 627–640.

Sullivan CA, Meigh J. 2007. Integration of the biophysical and social sciences using an indicator approach: Addressing water problems at different scales. Water Resources Management 21: 111–128.

Tognetti SS, Aylward B , Mendoza GF. 2005. Markets for watershed services. Page 193 in Anderson MG, ed. Encyclopedia of Hydrological Sciences. Wiley.

Tognetti SS, Aylward B, Bruijnzeel LA. 2011. Assessment needs to support the development of arrangements for payments for ecosystem services from tropical mountain cloud forests. Pages 671–685 in Bruijnzeel LA, Scatena FN, Hamilton LS, eds. Tropical Montane Cloud Forests: Science for Conservation and Management. Cambridge University Press.

Whittington D, Pagiola S. 2012. Design of payments for environmental services mechanisms: A review and assessment. The World Bank Research Observer 27: 261–287.

Wunder S. 2007. The efficiency of payments for environmental services in tropical conservation. Conservation Biology 21: 48–58.

Wunder S, Alban M. 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. Ecological Economics 65: 685–698.

Zhen N, Fu B, Lu Y, Zheng Z. 2014. Changes of livelihood due to land use shifts: A case study of Yanchang County in the Loess Plateau of China. Land Use Policy 40: 28–35.

Zheng H, Robinson BE, Liang Y-C, Polasky S, Ma D-C, Wang F-C, Ruckelshause M, Ouyang Z-Y, Daily GC. 2013. Benefits, costs, and livelihood implications of a regional payment for ecosystem service program. Proceedings of the National Academy of Sciences 110: 16681–16686.

Heidi Asbjornsen (heidi.asbjornsen@unh.edu) is affiliated with the Department of Natural Resources and the Environment and the Earth Systems Research Center, Institute for Earth, Oceans and Space, University of New Hampshire. Kathleen E. Halvorsen is affiliated with the Department of Social Sciences and School of Forest Resources and Environmental Science at Michigan Technological University, in Houghton. Alex S. Mayer is affiliated with the Department of Civil and Environmental Engineering at Michigan Technological University, in Houghton. Kelly W. Jones is affiliated with the Department of Human Dimensions of Natural Resources at Colorado State University, in Fort Collins. Theresa Selfa is affiliated with the Department of Environmental Studies, State University of New York, College of Environmental Science and Forestry, in Syracuse. Leonardo Saenz is affiliated with Conservation International, in Washington, DC. Randall K. Kolka is affiliated with the Northern Research Station of the US Department of Agriculture Forest Service, in Grand Rapids, Minnesota.