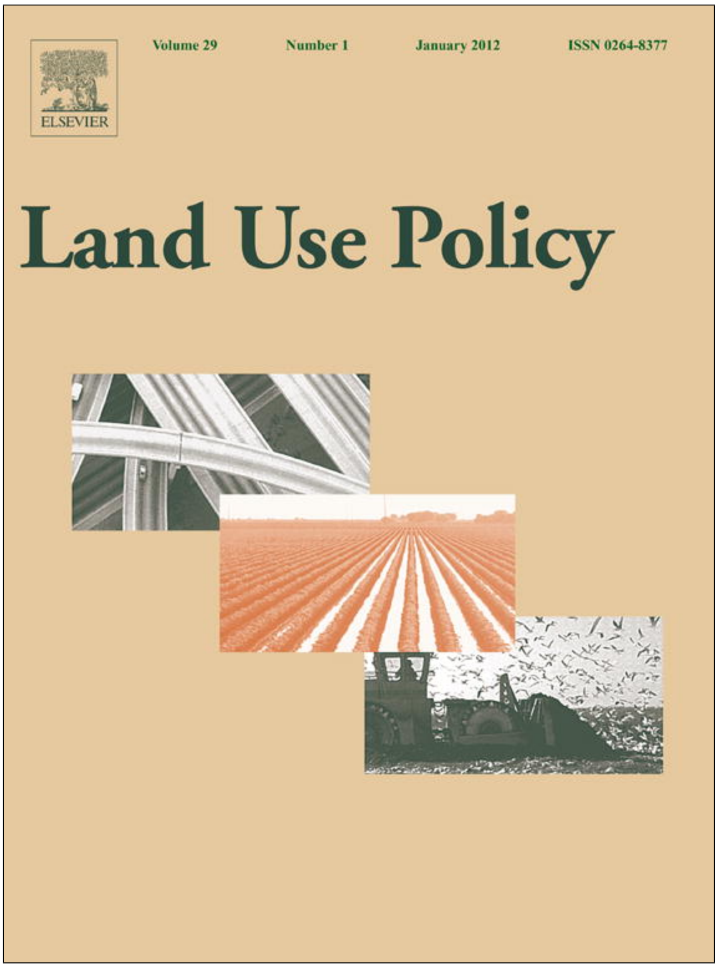
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Linking regional land use and payments for forest hydrological services: A case study of Hoa Binh Reservoir in Vietnam

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ABSTRACT

We have calculated the economic value of forest hydrological services for Hoa Binh Hydroelectric Plant in Vietnam, which is a major power supplier for the capital Hanoi. Our valuation is based on measurements over a six-year period from 2001 to 2006 in 240 permanent sample plots in different vegetation types distributed throughout the watershed. We have synthesized the information with GIS, and carried out simulations with derived empirical models for different land use, electricity price and payment proportion scenarios. Our findings indicate that the economic value of forest hydrological services for electricity production ranges from 26.3 million USD to 85.5 million USD per year; and that the longevity of the hydroelectric plant can be prolonged by about 35–80 years, depending on the state of forest cover in the watershed.

Introduction

The importance and advantage of forests in providing hydrological services are well known, and have been extensively documented (Chang, 2006; Börkey et al., 2005; Hewlett, 1982). Forest hydrological services are beneficial for hydroelectric production, where forests contribute to lower soil sedimentation and store water, and thus, maintain the capacity and prolong the longevity[1] of hydroelectric production plants (Rojas and Aylward, 2002; Nguyen and Vo, 1997). However, while it is clear that payments for ecosystem services (PES) are needed to internalize these positive externalities (García-Amado et al., 2011; Costanza et al., 1997; Coase, 1960), the basis for identifying the proper level of payments is under much discussion and substantially different from case to case (Wunder et al., 2008; Kosoy et al., 2007), creating difficulties for policy decision-making and practical application. Progress in the assessment of ecosystem services has been impeded by the lack of a standardized classification of which services to evaluate and how (Fisher et al., 2009; Boyd and Banzhaf, 2007). This is partly because it is often difficult to measure the output of ecosystem services. Furthermore, ecosystem services often have a "public goods" character which implies non-rivality and non-excludability, especially those from regulation services (Zander and Garnett, 2011; Daily et al., 2009). This leads to underestimation of service value, free-riding, undersupply, and finally, exploitation and environmental damage (TEEB, 2010).

Some of the key prevailing questions in PES schemes, thus, include: (1) who must pay? (2) who are paid? and (3) how much are the payments? Answering these questions can apparently make PES schemes more operational and practicable (Wunder, 2007; Balmford and Whitten, 2003). The quantification and valuation

Abbreviations: ES, Ecosystem Service(s); PES, Payment for Ecosystem Service(s); V_w, Payment for water provision service of forests; V_s, Payment for sediment prevention service of forests; V, *Total* payment for hydrological services of forests; p, *Electricity* price (VND/kWh); η, *Proportion* of increased electricity revenue paid to forest owners for water provision service; γ, *Proportion* of increased electricity revenue paid to forest owners for sediment prevention service; £, *Sediment* delivery ratio; R_p, Rainfall erosivity index; α, Slope (α,°); K, *Soil* erodibility index; Z, Vegetation index; DEM, Digital elevation map; PT, Annual rainfall (mm/year); LT, Annual throughfall (mm/year); MT, Annual stemflow (mm/year); LF, Annual water infiltrated and stored in literfall (mm/year); BM, Annual overland flow (mm/year); BH, Annual evapotranspiration during all rain events (mm/year); TT, Annual rainfall interception (mm/year); WI, Annual soil infiltrated water (mm/year); TH, Annual water taken by plants (mm/year); BD, Annual evaporation water from soil (mm/year); NN, Annual water flowing into the ground water (mm/year); WH, Annual water as soil moisture (mm/year); Ws, Annual water stored in soil (mm/year); CS, Normalized forest cover (%); $\hat{Y}$, Normalized forest area of the whole watershed; CT, Cover of forest trees (%); CS, Cover of shrubs (%); CG, Cover of grasses (%); CF, Cover of literfall (%); A, Soil eroded quantity (ton/ha/year); W1, Forest improvement by expanding forest area; W2, Forest improvement by increasing forest quality; VND, Vietnamese currency unit; USD, US dollar.

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[1] The longevity is herein understood as long as it is technically safe and operable (Wieland and Mueller, 2009).

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of ecosystem services are partly constrained by the disciplinary separation between ecological or environmental sciences and economics. The ecological underpinning of economic studies is often limited (Brookshire et al., 2007); and ecological models generally lack appropriate economic considerations (Brouwer and Hofkes, 2008). Obviously, integrating economics and ecological sciences into an operational decision support system is a key step required for global conservation and sustainability (Wei et al., 2009; Millennium Ecosystem Assessment, 2005).

In Vietnam, although forest hydrological services have been considered important (Water Resource Law 1998; Land Law 2003; Forest Protection and Development Law 2004), the legal framework of the payment for ecosystem services in general, and for forest hydrological services in particular, was established only in 2010 (see Wunder et al., 2005 for a review) with the promulgation of the Government's decree No. 99/2010/ND-CP (see Government of Vietnam, 2010). The decree stipulates that forest hydrological services exist that are advantageous for hydroelectric production, and that these must be rewarded. Thus, the quantification and valuation of those services must be carried out in order to establish a basis for the required payment. In this study we were motivated by three questions: (1) what is the economic value of forest hydrological services with respect to hydroelectric production? (2) since forests belong to different forest owners, how can one establish the level of payments for a specific forest stand? and (3) to what degree do potential land use changes influence the economic value in hydrological services that can be derived from a watershed? So far, we have focused our analysis on the most important forest hydrological services, namely for hydroelectric production via water storage and release (water provision), and in the prevention of soil loss with subsequent sedimentation of the reservoir (sediment prevention). Lower sedimentation plays an important feedback role in the economic system, since the longevity of the hydroelectric plant is prolonged. We applied our framework to the Hoa Binh Reservoir in the north of Vietnam, since the Hoa Binh Hydroelectric Plant and forest owners recently reached an agreement that the plant would pay the forest owners a certain proportion of the increased revenue for forest water provision and sediment prevention services. Our study is, thus, of practical significance for the implementation of this agreement. By extending our results to different land use change and electricity price scenarios, we hope that our findings will contribute useful information with respect to sustainable land use and formulation of forest management policy.

Literature review

PES are designed to provide economic compensation for the services ecosystems supply to society (see Elmqvist et al., 2010 for a review). PES systems must be both voluntary and contingent on the actual provision of ecosystem services (Pagiola, 2008). In order for PES to be implemented, ecosystem services must be identified and evaluated, and payment mechanisms must be established to encourage the provision of these services. Payments are normally given to landowners who implement or maintain desired land uses, which are thought to provide the ecosystem services of interest. In practice, most PES systems are "input-based", meaning that they compensate landowners for "inputs" such as trees planted, rather than for true "outputs" of ecosystem services such as, for example, increased biodiversity (Engel et al., 2008). This is because such outputs are difficult and expensive to assess and quantify.

Monetary value assigned to PES can in theory range from the opportunity costs to landowners to the true value of all ecosystem services provided, minus transaction costs. In reality, PES generally falls between these two extremes. For hydrological services, it is often assumed that the service user is the water use enterprise rather than the water end-user (Montagnini and Finney, 2011). In some cases, these enterprises finance their payments with additional fees levied on their end-users. However, in most cases, water use enterprises use their existing operating budget to make the payment (Pagiola and Platais, 2007). It is also quite often the case that, rather than evaluate, quantify, and monetarize actual ecosystem services provided, PES systems simply compensate landowners for provision cost. In this case, payments can be based on environmental targets and the cost to farmers for providing the desired land use (Pagiola et al., 2002). Obviously, this cost-covering compensation approach has several shortcomings. For example, it restricts the scope to those who bear some costs. Those who bear no costs do not need to be compensated. This is more problematic when service providers who suffer costs look not only for recompense, but also for a "provider surplus" – gains from the transaction that exceed their costs and make them better off (Wunder, 2007). An important characteristic of ecosystems and the services they provide is that they are not homogeneous across landscapes or seascapes, nor they are static phenomena (Fisher et al., 2009). Land use change and regional development clearly have implications for evaluation, quantification and monetarization of ecosystem services and vice versa (Rounsevell et al., 2010; Gren and Isacs, 2009). In this regard, the cost-covering compensation approach is even more disadvantageous.

The concept of ecosystem services is attracting increased attention as a way to communicate societal dependence on ecological life support systems (Turner and Daily, 2007; de Groot et al., 2002). Gómez-Baggethun et al. (2010) review the historic development of the conceptualization of ecosystem services and examine critical landmarks in economic theory and practice with regard to the incorporation of ecosystem services into markets and payment schemes. Daily and Matson (2008) highlight the tremendous value of ecosystem services and urge to turn this recognition into incentives and institutions that will guide wise investments in natural capital, featuring three key fronts: the science of ecosystem production functions and service mapping; the design of appropriate finance, policy, and governance systems; and the art of implementing these in diverse biophysical and social contexts. These arguments are supported by Daily et al. (2009) that we have not yet developed the scientific basis, nor the policy and finance mechanisms, for incorporating natural capital into resource- and land-use decisions on a large scale. Nevertheless some regional or local examples do exist. For example, Kosoy et al. (2007) compare three cases of payments for water-related ecosystem services in Central America based on opportunity costs of forest conservation and stakeholders' perceptions of the conditions on water resources and other issues. Branca et al. (2011) discuss how PES can lower the barriers for the adoption of sustainable land management practices in Tanzania. Zander and Garnett (2011) identify the economic value of ecosystem services on indigenous-held lands in Australia. If implemented properly, PES can be a tool for restoration and rural development. A number of studies have been devoted to a more practical question of how to make PES operationable. Muñoz-Piña et al., 2008 describe the process of policy design for PES in Mexico. Such studies are reviewed by Engel et al. (2008) where they state that PES is not a silver bullet that can be used to address any environmental problem, but a tool tailored to address a specific set of problems: those in which ecosystem services are mismanaged because many of their benefits are externalities from the perspective of ecosystem managers (Kinzig et al., 2011). Two important aspects of PES programs, namely the effectiveness and distributional implications, have also been considered, for example by García-Amado et al., 2011. Some authors have spent efforts to examine tradeoffs in ecosystem services and between conservation and development (Carreno et al., 2012; Raudsepp-Hearne et al., 2010; Rodríguez et al., 2006; Faith and Walker, 1996).

Even though ecosystem services research has become an important area of investigation over the past few decades, and the number of papers addressing ecosystem services is rising exponentially (Fisher et al., 2009), a very few number of papers have combined ecological measurements with economic valuations in forest hydrological service studies (see Johnston et al., 2011; Wei et al., 2009; Muñoz-Piña et al., 2008; Guo et al., 2000; Rosegrant et al., 2000; Aylward et al., 1998); and none of these directly deals with how to identify the payment for forest hydrological services. Our review thus indicates that the economic valuation of ecosystem services is still a challenge which requires a site-specific approach, as well as the integration of expertise from both ecological sciences and economics. This will be of particular interest to policy makers. In this regard we contribute to the current literature by (1) conceptualizing the linkage between forest hydrological services and the derived economic values, (2) identifying the economic value of and the level of payments for forest hydrological services, and (3) applying this framework to a case study in Vietnam – the Da River Watershed where Hoa Binh Hydroelectric Plant is located.

Conceptual approach

Our focus in the case of Hoa Binh Hydroelectric Plant includes water provision and sediment prevention services which are important for the operation of a hydroelectric plant. We assume that the economic value of services is the changed revenue of the plant due to the services provided, e.g., increased water provision and reduced soil sedimentation, leading to increased annual electricity production and prolonged life (longevity) of the reservoir.

Value of water provision by forests

We begin by assuming that the boundary of a watershed is clearly delineated, covering a specific land area which includes different land uses. This watershed provides water to a hydroelectric plant. We also assume that the plant agrees to pay forest owners, if there is an increased water supply for hydroelectric production due to forest maintenance. The quantity of water available for the plant, for example in one year, with and without forests is W_f and W_o, respectively. Thus, the change of water quantity per year would be:

$$\Delta W = W_f - W_o \tag{1}$$

Given the current technology of the plant, assume that β m^3 of water is needed to produce 1 kWh of electricity, ΔW will lead to the increased production of electricity per year (ΔE) as:

$$\Delta E = \frac{1}{\beta}(W_f - W_o) \tag{2}$$

If the price of electricity is p, the change of the revenue per year due to the increased water provision per year (ΔR_w) would be:

$$\Delta R_w = \Delta E \times p = \frac{1}{\beta}(W_f - W_o) \times p \tag{3}$$

Obviously this changed revenue is used to pay for the additional operation of the plant (i.e. labor, materials, etc.) and that only a certain proportion is paid to the service providers (forest owners). If an agreement is reached between the plant and the forest owners that the payment for water provision service is a certain percentage of the change, $\eta (0 \le \eta \le 100\%)$, then the monetary value of the water provision service would be:

$$V_w = \Delta R_w \times \eta = \frac{\eta}{\beta}(W_f - W_o) \times p \tag{4}$$

Thus, given η (as agreed between the forest owners and the plant), β (as constrained by the current technology of the plant) and p (as regulated by market or administrative mechanism), the key question to identify the economic value of water provision service of forests is to identify ΔW by comparing the quantity of water provided to the reservoir for hydroelectric production per year in cases with and without forests.

Value of sedimentation prevention by forests

Lowering the sediments accumulating in the reservoir will lead to two benefits, (1) increased longevity and, therefore, electrical power production at the dam, and (2) increased annual water storage capacity of the dam. While benefit 2 can be treated similarly to that described for water storage in forest soils (see section "Value of water provision by forests"), the following approach is used to describe benefit 1, the increased longevity of the dam due to the decreased sedimentation.

Sediment accumulated in the reservoir is due to soil erosion. Erosion and contribution of sediments vary due to rainfall, topography, soil properties and vegetation cover. Assume that given 1 m^3 of eroded soil, £ m^3 will be accumulated in the reservoir ($0 \le £ \le 1$). Assume also that the average quantity of eroded soil with and without forests per year is S_f (m^3/year) and S_o (m^3/year), respectively. The storage capacity of the dam as designed and built is M (m^3). The changed longevity (ΔT) of the dam is thus:

$$\Delta T = \frac{M}{(S_o - S_f) \times £} \tag{5}$$

Assume that the annual production of electricity of the plant is G, then the changed revenue (ΔR_s due to the changed longevity of the dam is:

$$\Delta R_s = \Delta T \times G \times p = \frac{M}{(S_o - S_f) \times £} \times G \times p \tag{6}$$

Similar to the case for water provision service, a certain proportion of the changed revenue due to the changed longevity will be paid to the forest owners, γ ($0 \le \gamma \le 100\%$). The economic value of sediment prevention service of forests would be:

$$V_s = \Delta R_s \times \gamma = \frac{M}{(S_o - S_f)\, £} \times G \times p \times \gamma \tag{7}$$

Given M and G (as designed and built), p (as regulated by market or administrative mechanism), γ (as agreed between the forest owners and the plant), the key question to identify the economic value of sediment prevention service of forests is to identify ΔS by comparing the quantity of soil eroded per year in cases with and without forests, and £, the sediment delivery ratio.

Payments provided for specific forest ecosystems

The total economic value (V) of forest hydrological services in our specific case which includes water provision and sediment prevention is identified as follows:

$$V = V_\mathrm{w} + V_\mathrm{s} \tag{8}$$

V could be considered the amount of payments that the hydroelectric plant pays to the forest owners. In other words, this is the transfer of total payments from the electricity sector to the forest sector. Therefore, we must identify the level of payment for a specific forest stand. In essence, we must find a way to allocate V among forest owners. If the total forest land area within the watershed is Y (ha). The average level of payments per ha of forest (F) for these two services is:

$$F = \frac{V}{Y} \tag{9}$$

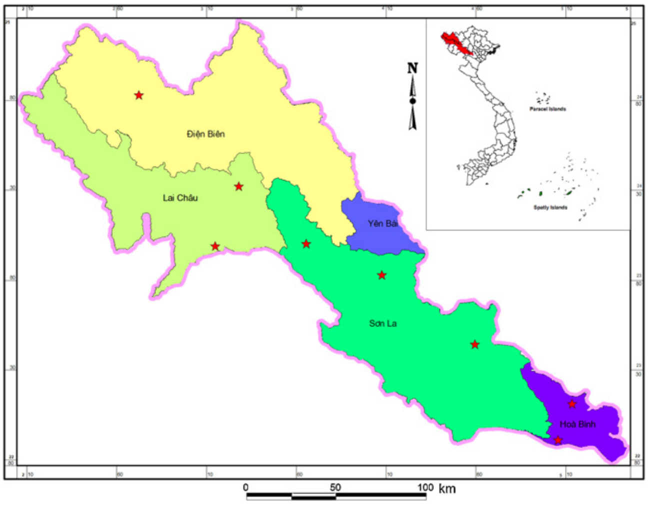

Fig. 1. Study area (the red stars indicate the districts where permanent sample plots were established along the river) (for interpretation of the references to color in this figure legend, the reader is referred to the web version of the article).

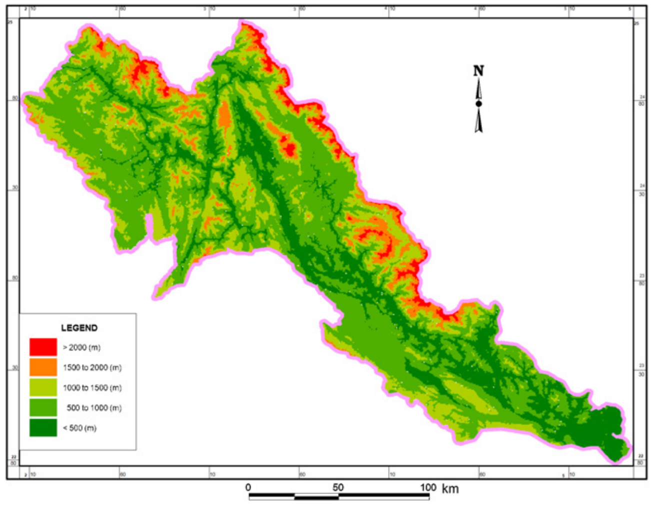

Fig. 2. Elevation map.

Obviously, different forest stands provide different quantities of water provision and sediment prevention services. The capacity of a forest stand to provide these services depends not only on its area but also on other factors. Therefore, we must adjust the area-based service estimated for a forest stand by taking into consideration actual ecosystem characteristics and properties (see Boyd and Banzhaf, 2007). In other words, we need to normalize or standardize this capacity of every forest stand. Thus, the level of payment for forest stand *i* is calculated as:

$$F_i = Y_i \times F \times C_i \tag{10}$$

where Y_i is the normalized forest area of forest stand *i*; *F* is the average level of payment; and *Ci* is the forest type adjustment factor. These are discussed further for the Da river watershed in the following section. It is noted that for the sake of simplicity to illustrate the conceptual framework, we describe here the linear relationship between the various factors. Thus, our study results should be interpreted with care.

Study site, data collection and modeling analysis

Site description

Hoa Binh Reservoir on the Da River is about 75 km west of Hanoi, Vietnam. The Da River flows from China via Vietnam to the East Sea. The length of the river in Vietnam's territory is 493 km and the average width is 1 km. The total surface area of the Da River Watershed is nearly 2.6 million ha in five provinces, namely Dien Bien, Lai Chau, Yen Bai, Son La, and Hoa Binh (Fig. 1).

The climate of the Da River Watershed is tropical monsoon with an average annual temperature from 22.5 to 23.2 °C. Annual precipitation ranges from 1300 to 2200 mm of which about 85% occur from May to September. The average annual humidity is high of 80–85%. The topography is complex with elevations from 300 to more than 2000 m above sea level. Only 19% of the land area have the elevations below 500 m; and 34% of the land area have the elevations higher than 1000 m (Fig. 2).

The complex topography is also illustrated with the various levels of land slopes. Only 3% of the land area have the slopes less than 10°; 54% of the land area have the slopes between 20 and 30°; and 12% of the land area have the slopes of more than 30% (Fig. 3). The downstream area of the Da River Watershed is the Red River Delta where Hanoi, the capital of Vietnam, is located. These indicate the importance of regulation of water and prevention of soil erosion in the study area.

The main soil type of the study area is Ferralsols (92%), including Rhodic Ferralsols, Xanthic Ferrasols, and Humic Ferralsols (Fig. 4) with the average initial topsoil (0–20 cm) contents of about 2% organic matter, 0.16% total nitrogen, 0.02% total P, 2% total K and 9.9 cmol per kg CEC. Average soil clay, silt and sand content are about 18, 29, and 53%, respectively (Dung et al., 2008). Soil infiltration rate varies much among soil and vegetation types. Under mature mixed plantations of *Pinus massoniana* and *Acacia mangium* on Humic Ferralsols, the initial soil infiltration rate ranges from 6.7 to 15.2 mm/min; the stable soil infiltration rate ranges from 2.5 to 8.0 mm/min (Pham, 2009). The time needed for the soil infiltration rate to be stable is from 50 to 125 min. For most upland soils, *K*sat is very low in the rainy reason, ranging from 65.7 cm/d in August to 421.2 cm/d in January in a cassava plot (Mai, 2007). Soil erosivity also varies from 0.06 to 0.30 depending on soil and vegetation types and average soil moisture before the rainy reason (in April) is from 7.37 to 10.38%.

There are different land uses in the watershed (Fig. 5). Grass and shrublands cover the largest share of the total land area, followed

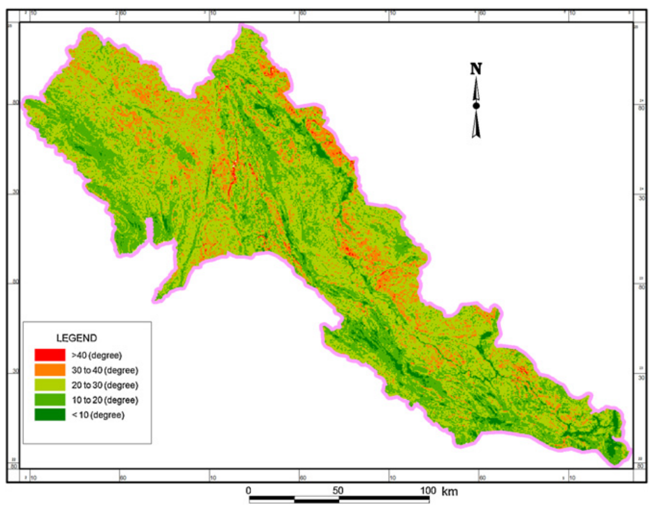

Fig. 3. Slope map.

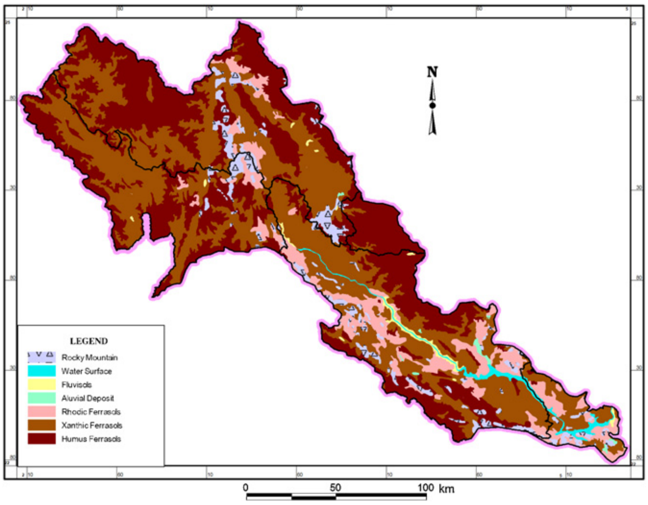

Fig. 4. Soil map.

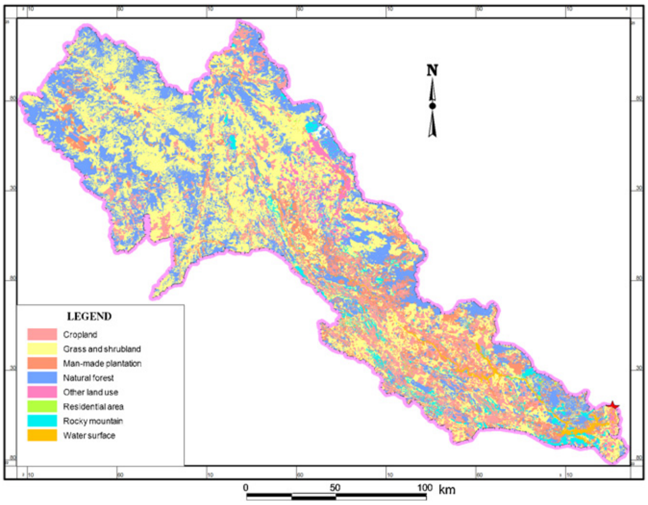

Fig. 5. Land use map.

Table 1
Current land uses in the entire Da river watershed (2009).

No	Land use type	Land area (1000 ha)	Land share (%)
1	Residential area	13.7	0.54
2	Water surface	21.1	0.82
3	Rocky mountain	71.8	2.80
4	Grass and shrubland	1083.4	42.32
5	Plantation	246.5	9.63
6	Natural forest	710.6	27.75
7	Agricultural cropland	362.8	14.17
8	Other land uses	50.4	1.97
Total		2560.3	100.00

by forests which include natural forests and plantations[2] (Table 1). However, natural forest distribution is fragmented, and consists mainly of secondary degraded natural forest stands (Nguyen et al., 2010). Agricultural cropland is mainly distributed in areas at low elevations where crop cultivation can take place (Nguyen, 2012).

Along the Da River, there are two hydroelectric plants, Hoa Binh and Son La. The construction of Hoa Binh Plant was begun in 1979 and completed in 1994 with a capacity of 1920 MW and an annual electricity production of 9 billion kWh. The construction of Son La Plant started in 2005 and is planned to be completed in 2014 with a designed capacity of 2400 MW and an annual electricity production of 9 billion kWh. Since Son La Plant is not yet completed, our valuation of forest hydrological services is only for Hoa Binh Plant. The maximum water level of Hoa Binh Dam is 120 m; and the dead water level is 80 m. The maximum water carrying capacity of the dam is 9.5 billion m^3. The total water discharge of the Dam is 49.75–50.80 billion m^3 per year of which about 35–36 billion m^3 are used for hydroelectric production. The water discharge varies substantially between dry and rainy seasons, from 1000 to 10,000 m^3 per second. The exhaustion of water in dry seasons leads to decreased electricity production. Thus, the provision of water for the plant in dry seasons is very critical, especially in the case of Vietnam where the shortage of electricity is currently considered one of the major constraints for economic growth (Nguyen and Dapice, 2010).

Modeling analysis

There are various methods currently used to quantify and valuate ecosystem services (Kumar and Kumar, 2008; National Academy of Sciences, 2005; Hawkins, 2003; Farber et al., 2002; Holling, 2001); and integration of economics with ecological modeling has been increasingly applied (Branca et al., 2011; Lin et al., 2007; Guo et al., 2000; Aylward et al., 1998). In situations where hydrological models have been combined with economic valuation methods, however, the degree of complexity in describing hydrological processes and the degree to which economic valuation is exercised is not always balanced or equally weighted (Kragt and Bennett, 2009; Rosegrant et al., 2000). Since our study is practically oriented, we focus on providing reliable recommendations for the payments ascribed to services, rather than pursuing a hydrological process description. Therefore, the treatment of erosion and water balance within the watershed was simplistic. Nevertheless, the illustrated principles will remain applicable even if alternative descriptions of the hydrological processes are used. Our procedure for the valuation of the two services included the following steps: (1) measurement in permanent sample plots to collect field data; (2) estimation of empirical regression models; and (3) derivation of payments and simulation of scenarios.

Measurement

We first considered that these two services provided by individual forest stands depend on four factors, namely rainfall, topography, soil properties, and vegetation cover (Pham, 2011; Bruijnzeel, 2004; Bonell, 1993; Douglas, 1977; Burger, 1954). These factors were quantified via a rainfall erosivity index (R_p, foot-tonf-inch/acre-hour-year)[3], land slope (α,°), soil erodibility index (K), and vegetation index (Z), respectively. In the watershed, based on available GIS maps (i.e. Digital Elevation Map (DEM), Land Use Map) and secondary data sources (Pham, 2009; Vuong, 2007; national and provincial statistics), we classified R_p into three levels (<500; 500–700; >700), α into five levels (<10; 10–20; 20–30; 30–40; >40), K into four levels (<0.12; 0.12–0.18; 0.18–0.24; >0.24), and Z into four types (grassland, shrubland, natural forests, and plantations). As shown in Table 1, these four vegetation types cover 80% of the study area. It means that the quantity of the services is compared among these vegetation types. Thus, in total we had $3 \times 5 \times 4 \times 4 = 240$ combinations of ecosystem descriptive factors. It is noted that these four factors are not completely independent. However, for such a complex study area with regard to various

[2] Plantation herein means man-made forest.

[3] For the conversion to SI metric system, see USDA (1997), pp. 325.

slopes, soil types, and vegetation characteristics, our experimental design was aimed to include these characteristics. Obviously, such a design is not scientifically perfect but can represent the study area at a certain level to provide reliable estimates of the relationship between these various factors that need to be measured and our variables of interest, namely water storage and soil erosion. For each of the combinations, we established one experimental field plot (400 m^2, 20 m × 20 m) to collect our data on soil loss, components of water balance, characteristics of soil, litterfall, and other variables. On each sample plot, the following measurements were conducted.

– The components of water balance were measured with 36 rain events and then calculated in a year, including (1) the amount of rainfall (PT, mm/year), (2) the amount of throughfall (LT, mm/year), (3) the amount of stemflow (MT, mm/year), (4) the amount of water infiltrated and then stored in litterfall (LF, mm/year), (5) the amount of overland flow (BM, mm/year), (6) the amount of evapotranspiration during all rain events (BH, mm/year), and (7) the amount of rainfall interception (TT, mm/year) which is kept in the canopy of trees. The measurements followed Hewlett (1982) and were described in more detail by Pham (2011). These data were used to identify the water balance and the rainfall erosivity index (R_p).
– The slope (α) was identified from the slope map and validated with the field measurements of five points within the plot. The slope of a plot is the mean slope of all pixels of the plot. Each pixel is 30 m × 30 m.
– The characteristics of a vegetation type were measured, including the number and canopy of trees, the cover of trees (CT), shrubs (CS), grass (CG), and literfall (CF). These data were used to identify the vegetation index (Z). Within each sample plot, we also established five secondary quads (1 m^2, 1 m × 1 m) to measure the amount of litterfall. Literfall was then put into water in 30 min, 1 h, 4 h, and 24 h to identify its average water absorption and retention capacities.
– The characteristics of soil were measured, including (1) total percentage of silt and very fine sand, (2) percentage of sand of 0.10–2.0 mm, (3) percentage of soil organic matter, (4) soil structure, (5) soil permeability. These data were used to identify the soil erodibility index (K) following Wischmeier and Smith (1978). We also measured water infiltration of soil. Soil samples were taken at the depth of 0–10, 10–30, and 30–60 cm and analyzed at the Soil Lab of Forestry University of Vietnam. The water amount of soil transpiration was identified by the gravimetric method. The water amount of evaporation of soil was identified by comparing the weight of soil samples after a time interval of 5 min (for a more detailed description of the measurements, see Pham, 2011, 2009; Vuong, 2011; MARD, 2010). Our measurements were conducted over a six-year period from 2001 to 2006.

Estimation of empirical regression models

One of our tasks was to identify the amount of overland flow during the rainy reason, and the annual water retaining capacity of the soil. Part of the overland flow in the rainy season and all soil-retained water are used for hydroelectric production. We followed Vuong (2007) that 20% of the overland flow in the rainy season is used for hydroelectric production. It was obvious that the water balance was needed in order to calculate water retaining capacity of a vegetation type in a year.

It is well known that hillslope hydrological processes at forest stand scale are very complex, including various processes and their interactions depending on spatial and temporal dimensions. However, at the time interval of one year, it is possible to simplify these processes to derive a reasonable estimate of water balance components. We thus followed Chang (2006), Bruijnzeel (1990) and Douglas (1977) to identify the water balance and water retaining capacity of a vegetation type in a year as follows:

$$\mathrm{PT} = \mathrm{TT} + \mathrm{BH} + \mathrm{LT} + \mathrm{MT} \tag{11}$$

Similar to BH, TT will be evaporated into the atmosphere as the so-called interception loss. LT and MT will go to the ground floor and are divided into three components: (1) water absorbed and kept by literfall (LF), (2) overland flow (BM), and (3) soil infiltrated water (WI). Thus, Equation (11) becomes:

$$\mathrm{PT} = \mathrm{TT} + \mathrm{BH} + \mathrm{LF} + \mathrm{BM} + \mathrm{WI} \tag{12}$$

The process of water infiltration into soils is very complex but quantitatively WI (mm/year) can be identified as:

$$\mathrm{WI} = \mathrm{TH} + \mathrm{BD} + \mathrm{NM} + \mathrm{WH} \tag{13}$$

where TH is the amount of water taken by plants, BD is the amount of water evaporation from soil, NN is the flow to ground water (mm/year), WH is the amount of water remaining as soil moisture (mm/year). These two last components (NN and WH) are part of water resources provided for hydroelectric production (in addition to 20% of overland flow) and summed up as W_S.

$$W_S = \mathrm{NM} + \mathrm{WH} \tag{14}$$

Thus, W_S (mm/year) is the amount of water stored on an annual time step belowground (mm/year) and can be calculated from the measurements of water balance components.

Next, from the experimental data and the characteristics of the factors described above, we derived empirical regression models (Chiang, 2003) for the relationships between the percentage of overland flow water (BM, % per total rainfall/year), percentage of soil-retained water (W_S, % per total rainfall/year), and the amount of soil loss (A, tons/ha/year). The independent variables in these regression models include R_p, α, K, and Z. The vegetation index Z (%) for each vegetation type was calculated as $Z = \mathrm{CT} + \mathrm{CS} + \mathrm{CG} + \mathrm{CF}$ where CT (%) is the cover of forest trees; CS (%) is the cover of shrubs; CG (%) is the cover of grasses and CF (%) is the cover of literfall. It is noted that CT is calculated for both natural forest and plantation stands. The empirical models were selected based on (1) the non-linear relationship between the dependent and independent variables, and (2) the highest value of R^2, and (3) the derived curves follow the trend of observed data distribution.

Then we used the relevant maps (slope map, vegetation map, etc.) to spatially distribute the parameters and estimates. The experimentally derived empirical models are the backbone of our analysis at the watershed scale in order to quantify hydrological services under different rainfall erosivities, land slopes, soil erodibilities, and vegetation types. Thus, we were able to estimate the changes in water provision per year (m^3/year) and soil loss per year (tons/year) associated with different vegetation types distributed within the watershed.

The capacity for water storage and soil loss prevention is different between vegetation types and depends, as shown, also on other factors. Thus, it is not possible to compare this capacity among vegetation types with, for example, different slopes and soil types. Therefore, we normalized across vegetation types via the soil loss prevention capacity. Any soil loss quantity can actually be used as "the standard reference value" in this procedure. Our experiments showed that the average soil loss per ha per year varies; and the soil loss under natural rich forests is the lowest (from 4.5 to 6.5 tons/ha/year). Therefore, we chose a soil loss of 5.5 tons/ha/year as the standard reference value. Consequently, the normalized area ($\hat{y}$, ha) of a ha of forest stand i is calculated as $\hat{y}_i = \frac{5.5}{A_i}$, where A_i is the soil loss per ha per year of that forest stand i. Thus, the total normalized area ($\hat{Y}$, ha) of the whole watershed was calculated as

Table 2
Annual average water balance and soil erosion of different vegetation types.

	Grassland	Shrubland	Natural forest	Plantation
PT (mm/year)	2005.7 (129.86)	2005.7 (130.1)	2005.7 (126.6)	2005.7 (127.16)
BT (mm/year)	58.4 (4.57)	174.5 (33.16)	321.2 (44.55)	204.2 (21.65)
LT (mm/year)	1947.3 (129.86)	1831.2 (99.31)	1589 (117.15)	1736.1 (116.93)
MT (mm/year	0	0	95.5 (12.03)	65.4 (7.07)
LF (mm/year)	2.5 (0.21)	26.4 (2.2)	46.1 (3.67)	67.9 (4.85)
BM (mm/year)	575 (90.48)	320 (234.26)	137 (28.49)	191 (89.77)
WI (mm/ha/year)	1369.8 (72.56)	1484.8 (232.96)	1501.4 (292.11)	1542.6 (194.66)
TH (mm/year)	236 (11.99)	364 (30.87)	483 (53.53)	505 (35.73)
BH (mm/year)	755 (76.83)	534 (165.84)	420 (250.78)	471 (182.05)
WS (mm/year)	378.8 (33.66)	586.8 (80.69)	598.4 (78.69)	565.6 (48.56)
A (tons/ha/year)	24.88 (13.69)	20.67 (5.54)	6.47 (6.08)	10.19 (2.82)

(Standard deviation in parentheses).

$\hat{Y} = \sum_{i=1}^{m} \hat{y}_i \times y_i$ where y_i is the real area of forest stand i and m is the number of forest stands. In this way, the current forest cover in the whole watershed was actually equal to the normalized forest cover (CS, %) of only 30.8%. The normalized forest cover is the important indicator when we consider future land use scenarios where forests are enriched or degraded in the next section.

Derivation of payments and simulation of scenarios

Following Vuong (2011, 2007), the sediment delivery ratio was identified based also on the level of influence which includes direct or indirect influence on water regulation and soil erosion (Fig. 6). This is to take into account the spatial variability of the service provision of a forest stand. Thus, within the watershed, the sediment delivery ratio was 0.67 for the areas that have a direct influence and 0.48 for the areas that have an indirect influence.

Next, we considered how much the payment should be in practice for a specific forest stand. Therefore, if V is the total payment for a service, the payment for a ha of normalized forest was calculated as $\frac{V}{\hat{Y}}$. The payment for each forest stand will depend on its normalized forest area. It means that two forest stands with the same real area may have different levels of payments due to their different normalized forest areas.

We then simulated the quantity of the services with regard to changes in (1) land use (in terms of the normalized forest cover CS), (2) electricity price (p); and (3) the proportion of changed revenue paid to forest owners (η and γ). In terms of land use change, the following scenarios were considered: (i) baseline scenario with the current land use of 2009 (CS = 30.8%); (ii) without-forest scenario where current forests are converted to grass- or shrublands

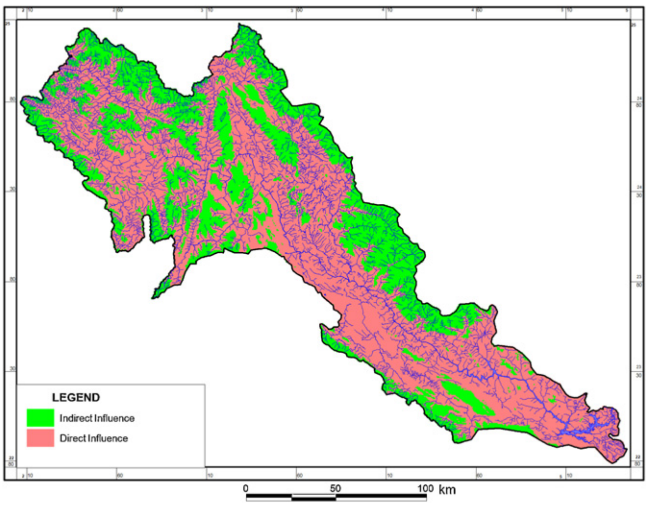

Fig. 6. Influence level map.

Table 3
Quantity of eroded soil in the entire watershed with different CS.

CS (%)	Eroded soil (million tons/year)		
	Mean	Max.	Min.
0.00	51.8	56.5	47.1
30.80	34.5	37.6	31.4
35.00	32.7	35.7	29.7
40.00	30.6	33.4	27.8
45.00	28.4	31.0	25.8
50.00	26.3	28.7	23.9
55.00	24.1	26.3	21.9

(CS = 0%). This scenario is in fact relevant, because shifting cultivation (slash and burn) still exists in the watershed (Nguyen et al., 2010); and (ii) an improved-forest scenario (CS increases to 35%, 40%, 45%, 50%, and 55%). This scenario is also very relevant, since the importance of forests in the watershed has been realized and payments for hydrological services are under study. The increase of the normalized forest cover can be implemented in two ways: (1) by expanding the current forest area (W1), or (2) by increasing the quality of the current forest area (W2).

With regard to the electricity price, in 2009 the government of Vietnam regulated that the price was 1000 VND[4] per kWh. It is noted that the Vietnam Electricity Corporation (EVN) is the governmental company. However, EVN is claiming that it is suffering from losses due to this too low electricity price and declared that the production cost per kWh is 1180 VND. Thus, we performed our economic valuation with three electricity prices per kWh: (i) 1000 VND (as it is now), (ii) 1180 VND (as the production cost declared by the EVN), and (iii) 1300 VND (as about 10% increase of the production cost so that the EVN has a certain profit from production).

Lastly, after various negotiation efforts, an agreement is reached between Hoa Binh Hydroelectric Plant and forest owners in the watershed that the plant would pay 40% and 10% of its increases of hydroelectric revenue for water provision and sediment prevention services, respectively (η = 40% and γ = 10%). However, there have been requests from forest owners to increase these proportions of payments. Thus, we considered two schemes of the payment proportions: (i) 40% and 10%; and (ii) 50% and 20% (η = 50% and γ = 20%) for water provision and sediment prevention services.

Results and discussion

The annual quantities of water balance components and soil erosion of different vegetation types in the study area were identified (Table 2). With the average annual rainfall of 2005.7 mm, the

[4] VND is the abbreviation of Vietnam Dong, the currency unit of Vietnam. 1 US$ equaled about 20,000 VND in 2010.

Table 4
Changes of the services with different CS for the entire watershed.

Parameters	Dimension	CS (%)					
		30.80	35.00	40.00	45.00	50.00	55.00
ΔBM	Million m³/year	842	920	1015	1109	1204	1298
ΔW_S	Million m³/year	2112	2309	2546	2783	3021	3258
ΔSD	Million tons/year	17.3	19.9	21.2	23.4	25.5	27.7

amount of throughfall is the highest in grassland whereas in plantations the amount of rainfall interception is the highest. Among these vegetation types, natural forest stands have the highest water storage capacity; however, plantations have a lower quantity of eroded soil in comparison with natural forests. This is because natural forests in the study area are mainly secondary and degraded. Soil erosion is much more serious in grassland and shrubland (further degraded). It is obvious that the capacity of forests (both natural forests and plantations) for water storage and soil erosion prevention is higher than that of other existing vegetation types. Our findings are consistent with many other authors, e.g. Calder (2007); Bishop and Landell-Mills (2002); Hamilton and Pearce (1986) who show that forests are comparatively advantageous in water and soil regulation than other vegetation types.

From the data measured in the sample plots, our derived regression models for the relationships between our dependent variables (BM, W_S, and A) and the influencing factors were constructed as follows:

$$\mathrm{BM} = 0.1624 \times R_p \times \left(Z/(K \times \alpha)\right)^{-0.576} \quad \text{with} \quad R^2 = 0.90 \qquad (15)$$

$$W_S = 31.117 - 0.366 \times R_p/\left(\frac{Z}{K \times \alpha}\right) \quad \text{with} \quad R^2 = 0.81 \qquad (16)$$

$$A = 0.231 \times R_p \times \left(Z/(K \times \alpha)\right)^{-0.833} \quad \text{with} \quad R^2 = 0.80 \qquad (17)$$

where BM and W_S are measured as the percentage of the annual total rainfall (%); A is measured in ton per ha per year.

BM and W_S were estimated for all cases of R_p, but for simplification A is illustrated in Fig. 7 only for R_p of 800 foot-tons/acre/year (in the two other cases of R_p of 600 and 400 foot-tons/acre/year, the curves are similar in form but below the A curve for R_p of 800). This figure indicates that, BM and W_S are negatively correlated. If the cover of the vegetation is low but other factors remain the same, then the amount of overland flow is high, and the water retention capacity of the soil is also low. Similarly, the annual amount of eroded soil and overland flow is low when the cover of the vegetation is high. This demonstrates that the model describes the role of forest stands in water flow regulation and soil erosion control in a reasonable fashion.

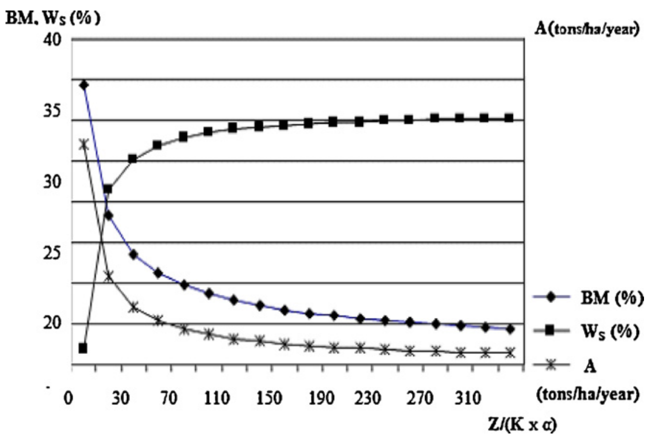

Fig. 7. Relationship between BM, W_S, and A with the influencing factors.

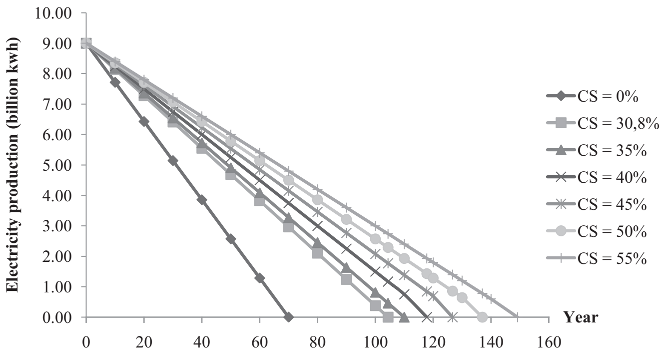

Fig. 8. Hydroelectric production and longevity of the dam with different CS.

From these empirical models, we stimulated the eroded soil quantity for the previously described land use scenarios. The results showed that for the whole watershed, the annual amount of eroded soil in the "without-forest scenario" would be highest, about two times more than that of the "improved-forest scenario" with the normalized forest cover of 50% (Table 3).

Lower levels of soil erosion lead to less sediment accumulation in the reservoir, and consequently enhance the longevity of the reservoir. Thus, with the normalized forest cover (CS) of 0% (without-forest scenario), 30.8% (baseline scenario), 35%, 40%, 45%, 50%, and 55% (improved-forest scenario), the usefulness of the dam would be 69.5 years, 104.4 years, 110 years, 117.8 years, 126.7 years, 137 years, and 149.2 years, respectively (Fig. 8). Forests can contribute to an increase in longevity of the dam by 34.9–79.7 years, depending on the extent of their forest cover.

Table 4 describes in terms of ΔBM (the decrease in overland flow), ΔW_S (the increase in soil-retained water), and ΔSD (the decrease in soil sedimentation) the services derived in the improved-forest scenario as compared to the scenario without-forest. The values confirm our earlier statement that forests are advantageous in providing the desired services and, thus, the service providers must be paid.

Based on the quantification of the services and the agreement between Hoa Binh Hydroelectric Plant with the forest owners, we were able to determine the payments for these services as shown in Table 5 for the base land use scenario. With the current price of hydroelectricity of 1000 VND/kWh,[5] the total payment for the forest hydrological services in the whole watershed was calculated as 578.3 billion VND/year (or 26.3 million USD/year). Thus, the payment for each ha of normalized forest and for each kWh of electricity was 733,000 VND and 78.4 VND, respectively. This indicates that the payment for hydrological services is about 7.84% of the current price of electricity. This finding is important for policy makers because due to electricity shortage, there is a currently a strong discussion and call to increase the electricity price. This should promote more investment from private sectors in electricity production. Future policy formulation for hydroelectricity may

[5] Note that this price is regulated by the Government of Vietnam.

Table 5
Quantity and monetary value of forest hydrological services in the base land use scenario with the current electricity price of 1000 VND.

No.	Parameter	Value
1	Water increase for hydroelectricity production (million m^3/year)	1270
2	Hydroelectricity increase due to the water provision service (million kWh/year)	318
3	Current price of hydroelectricity (VND/kWh)	1000
4	Revenue increase due to the water provision service (million VND)	317,700
5	Payment for the water provision service (40% of the revenue increase) (Million VND)	127,100
6	Average payment for the water provision service per ha of real forests (VND/ha/year)	126,000
7	Payment for the water provision service per ha of normalized forest (VND/ha/year)	161,000
8	Payment for the water provision service per kWh (VND/kWh)	14
9	Reduce of sediments in the dam (million tons)	17.3
10	Increase of the longevity of the reservoir (years)	34.9
11	Increase of hydroelectricity production due to the sediment prevention service (million kWh/year)	4,513,000
12	Revenue increase due to the sediment prevention service (million VND)	4,512,900
13	Payment for the sediment prevention service (10% of the revenue increase) (Million VND)	451,300
14	Average payment for the sediment prevention service per ha of real forests (VND/ha/year)	449,000
15	Payment for the sediment prevention service per ha of normalized forests (VND/ha/year)	572,000
16	Payment for the sediment prevention service per kWh (VND/kWh)	64

Table 6
Payments for forest hydrological services per kWh electricity with the current electricity price of 1000 VND.

CS (%)	Payment for the forest hydrological services (VND/kWh)			
	Water provision	Sediment prevention	Total	Change (%)
30.80	14.10	64.30	78.40	
35.00	16.00	73.00	89.00	13.53
40.00	18.30	83.40	101.70	14.27
45.00	20.60	93.80	114.40	12.49
50.00	22.90	104.20	127.10	11.11
55.00	25.10	114.60	139.70	10.00

Table 7
Average payments for forest hydrological services per ha of real forest with the current electricity price of 1000 VND.

CS (%)	W1 (1000 VND/ha of real forest)			W2 (1000 VND/ha of real forest)		
	Water provision	Sediment prevention	Total	Water provision	Sediment prevention	Total
30.80	126.30	448.60	574.90	126.30	448.60	574.90
35.00	127.00	412.60	539.60	143.30	448.60	591.90
40.00	127.70	379.30	507.00	163.80	452.30	616.10
45.00	128.30	351.20	479.50	184.30	455.60	639.90
50.00	128.90	326.70	455.60	204.70	457.90	662.60
55.00	129.20	305.60	434.80	225.40	460.80	686.20

need to take into account the payment for hydrological services provided by forests.

To facilitate the payment from the hydroelectric sector to the forest sector in the base and improved-forest scenarios, the payment for each kWh of the increase in hydroelectric production was calculated (Table 6).

As previously described, in the improved-forest scenario there are two ways to increase the normalized forest cover, either by increasing forest area (W1) or by improving the quality of the current forests (W2). These two ways lead to different levels of payment for each ha of real forest (Table 7). If W1 is selected, the average payment per ha of real forest would decrease even though the total economic value of the services in the whole watershed and the longevity of the reservoir would increase. This is because the total economic value of the services increases more slowly than the increase in forest area, leading to the decreased average payment per ha. If W2 is selected, then the average payment per ha would increase. This suggests that forest owners should enrich the current forests in order to get higher levels of payments for the services per ha of forest.

Table 8 describes the total annual payment of the services of forests for the whole watershed in the base land use scenario (CS = 30.8%) but with different electricity prices and payment proportions. It indicates that the economic value of forest hydrological

Table 8
Total annual economic value of the services in the base land use with different electricity prices and payment proportion.

Electricity price (VND/kWh)	η = 40% and γ = 10%			η = 50% and γ = 20%		
	Water provision (million VND/year)	Sediment prevention (million VND/year)	Total (million VND/year)	Water provision (million VND/year)	Sediment prevention (million VND/year)	Total (million VND/year)
1000	127,100	451,300	578,400	317,750	1,128,250	1,446,000
1180	149,978	532,534	682,512	374,945	1,331,335	1,706,280
1300	165,230	586,690	751,920	413,075	1,466,725	1,879,800

services for electricity production would be about 85.5 million USD if electricity price increases to 1300 VND/kWh as requested by the EVN and η = 50% and γ = 20%.

Conclusions and policy implications

A long history exists describing the interest of hydrologists and ecosystem scientists in the relationships between land use and the dynamics of regional water cycles (Engler, 1919; Bates and Henry, 1928). Conversions from forest to grass- or shrublands, variations in forest cover and other factors have been documented in terms of their influences on ecosystem services. In the work described here, we have focused on relating the main inputs (water volume storage in the reservoir and forest soils along with reservoir sedimentation) and a single important output (water volume available for electrical power generation) to patterns in land use within a monsoon climate hydrological reservoir system of Vietnam. We have built an initial tie to the social–ecological system by considering the economics of both sedimentation (losses due to ecosystem exports) and power generation (gains via overall water storage). We combined experiments with data manipulation and model simulations to identify the economic value of the two main forest hydrological services for hydroelectric production in the Hoa Binh Reservoir, Vietnam. The empirical models between the dependent variables of the percentage of overland flow water, the percentage of soil-retained water, and the quantity of eroded soil with the independent variables of rainfall erosivity, land slope, soil erodibility, and vegetation types were constructed from our experimental data over the time period 2001–2006. We simulated our results with three different land use scenarios, namely baseline scenario of land use 2009, without-forest scenario, and improved-forest scenario (including the increase of the normalized forest cover to 35, 40, 45, 50, and 55%), with different electricity prices (1000, 1180, and 1300 VND per kWh), and with different payment proportions paid to forest owners (40% of water provision service and 10% of sediment prevention service or 50% of water provision service and 20% sediment prevention service). Our main findings are summarized as follows.

First, it was found that with the current normalized forest cover of 30.8% and current electricity price of 1000 VND/kWh, total monetary value of these hydrological services is 578.3 billion VND (26.3 million USD) per year of which about 22% is for the water provision service and the rest is for the sediment prevention service. The payment per year for each ha of normalized forest and for each kWh of electricity was 733,000 VND and 78.4 VND, respectively. It is noted that the payment of those services per kWh per year is currently equivalent to 7.84% of the electricity price.

Second, the payment for these services per year would change if the normalized forest cover changes. The payment per year per kWh of electricity can be as high of about 14% of the electricity price if the normalized forest cover is of 55%. Similarly, the payment for these services would change if the electricity price and the revenue proportions paid to forest owners are changed. If the electricity price is 1300 VND/kWh and the revenue proportion paid to forest owners are 50:20 (η = 50% and γ = 20%), the payment per year for forests of the whole watershed would be about 85.5 million USD. Future policy formulation for hydroelectricity may need to take into account the payment for hydrological services provided by forests.

Third, there are two ways of increasing the normalized forest cover in the study area. Farmers may either expand the current forest area or improve the current forest quality. We concluded that the second way should be preferred as it increases the level of payments for forest owners. To accomplish this, we suggest that it is critically important to promote better relationships in resource management that allow coordinated efforts by ecosystem scientists, hydrologists, water managers and stakeholders.

Our analysis can be enriched in a number of ways. Our models are empirical and static. This obviously leaves some room for further investigation. Application of different soil and water assessment models, for example SWAT, will be themes of our future studies to provide more reliable estimates of the services. Moreover, mechanisms for the implementation of PES and associated transaction cost must also be examined. Such issues have not been studied adequately in Vietnam.

Successful management of watershed natural resources within a reservoir system requires a deep understanding of ecosystems and possible land use and policy changes. Developing the understanding described above is a challenging task, since it depends on climate, the properties of many types of ecosystems, spatial distribution of ecosystem types across the topography of watersheds, and the management and policy measures imposed within the context of a specific social–ecological system. We propose that the required understanding can be achieved in a stepwise procedure which evaluates the relationships among key ecosystem services from different perspectives. A stepwise analysis such as that undertaken here provides a basic economic framework within which a broader economic evaluation of services can be carried out.

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