ECOLOGICAL ECONOMICS 65 (2008) 48–62

available at www.sciencedirect.com

ScienceDirect

www.elsevier.com/locate/ecolecon

METHODS

Confronting unfamiliarity with ecosystem functions: The case for an ecosystem service approach to environmental valuation with stated preference methods

J. Barkmann[a,*], K. Glenk[a,1], A. Keil[a,2], C. Leemhuis[b,3], N. Dietrich[c,4], G. Gerold[b], R. Marggraf[a]

[a]Department of Agricultural Economics and Rural Development, Environmental & Resource Economics, Georg-August-Universität Göttingen, D-37073 Göttingen, Germany
[b]Institute of Geography, Landscape Ecology, Georg-August-Universität Göttingen, D-37077 Göttingen, Germany
[c]Institute of Geography, Social and Cultural Geography, Georg-August-Universität Göttingen, D-37077 Göttingen, Germany

ARTICLE INFO

Article history:
Received 9 January 2006
Received in revised form
18 November 2007
Accepted 2 December 2007
Available online 14 January 2008

Keywords:
Ecosystem services
Ecosystem functions
Willingness-to-pay
Stated preference methods
Information and
misspecification bias
Data requirements

ABSTRACT

Ecosystem functions are a central topic of environmental valuation research. Lay respondents are usually unfamiliar with the implications of scientific descriptions of ecosystem functioning. Thus, the applicability of stated preference methods for the valuation of ecosystem functions is a matter of debate. In the general discourse on the economic valuation of ecosystem functions, it was suggested to valuate ecosystem functions via the ecosystem services they provide. In this contribution, we argue that the recognition of this principle is the key also for the applicability of stated preference methods to the valuation of ecosystem functions. Successful application requires a precise differentiation between the descriptive realm of ecosystem functions and the evaluative realm of ecosystem services. On this basis, an ecosystem service approach for the economic valuation of ecosystem functions is introduced.

We illustrate the ecosystem service approach by the valuation of a hydrological ecosystem function in rural Indonesia. Identification and representation of the ecosystem services were based on extensive investigations of respondent perception of hydrological phenomena. The availability of irrigation water during the dry season turned out to be a locally decisive ecosystem service. Within the case study, willingness-to-pay (WTP) values were estimated for changes in the availability of irrigation water, rattan, shading in cacao plantations, and in the population of an endemic mammal by Nested Logit (NL) analysis of choice experiment data. Only few respondents lacked sufficient familiarity of the

* *Corresponding author*. Department of Agricultural Economics and Rural Development, Environmental and Resource Economics, Georg-August-Universität Göttingen, Platz der Göttinger Sieben 5, D-37073 Göttingen, Germany. Tel.: +49 551 39 14492; fax: +49 551 39 14812.
E-mail address: jbarkma@gwdg.de (J. Barkmann).
[1] Present address: Socio-Economics Group, Macaulay Institute, Craigiebuckler, Aberdeen, AB15 8QH UK.
[2] Present address: Institute of Agricultural Economics and Social Sciences in the Tropics and Subtropics, University of Hohenheim, D-70593 Stuttgart, Germany.
[3] Present address: Center for Development Research (ZEF), Div. Ecology and Natural Resources, Walter-Flex-Str. 3, D-53113 Bonn, Germany.
[4] Present address: International Hydrological Programme, Division of Water Sciences UNESCO, 1 rue Miollis, F-75732 Paris Cedex 15, France.

doi:10.1016/j.ecolecon.2007.12.002

environmental goods to be valued. WTP for a reduction in water scarcity by one month was ~39–40,000 Indonesian Rupiah/household/year. A rating of respondent comprehension correlates with preferences for water availability. For respondents with above average comprehension (rated 4 versus mean of 3.12 at a 5-point scale), NL analysis predicts a 14.1% increase in WTP. Interactions of attitudinal constructs support the notion that the WTP estimate is an expression of underlying values and risk perceptions.
The ecosystem service approach comes at a cost. Because ecosystem services relevant to local respondents are valued rather than scientifically described ecosystem functions, typical 'basic science' models that represent ecosystem functioning cannot be used for the analysis of valuation scenarios with direct policy relevance. Engineering-type models that embody technical and, in our case study, agronomic knowledge are necessary to bridge the gap between ecosystem functioning and their practical implications. A lack of this kind of information also hinders a meaningful application of alternative valuation approaches.

1. Introduction

Ecosystem functions and the benefits humans derive from them have become a central topic of research at the interface of social and natural systems (Costanza et al., 1997; Daily, 1997; Carpenter and Turner, 2000; Farber et al., 2002; Alcamo et al., 2003; Heal et al., 2005). Along with the recognition of their economic importance, there is an intensive debate on the economic valuation of ecosystem functions and the services they provide (Costanza and Farber, 2002).

In particular, there is substantial disagreement on the suitability of stated preference methods (contingent valuation, choice modeling) for the economic valuation of ecosystem functions.[5] The Millennium Ecosystem Assessment, for example, relies on contingent valuation as a "commonly used" valuation method for the quantification of indirect use values to which ecosystem functions belong (Alcamo et al., 2003). In contrast, de Groot et al. (2002, p. 404; Table 2) suggest to restrict stated preference methods to a valuation of "information services", such as ecosystem services for recreation and tourism. While the German Council of Environmental Advisors demands non-optimizing "categorical" valuation methods for "essential" ecosystem functions (WBGU, 1999), stated preference practitioners regularly include ecosystem functions along with other passive use values in the set of suitable objects for stated preference studies (e.g., Carson et al., 1999).

Echoing earlier critical assessments (e.g., Diamond and Hausman, 1994), one particularly serious challenge for the applicability of stated preference methods to ecosystem functions was put forward by Nunes and Bergh (2001). They stress that lay respondents surveyed in stated preference studies will usually lack sufficient insight into ecosystem life support functions and processes such as photosynthesis or biogeochemical matter cycling. Without sufficient familiarity with ecosystem functions, respondents are not able to make meaningful preference statements. Although the critique is well taken with regard to the valuation of ecosystem *functions*, it is less clear, however, that the critique also applies to the ecosystem *services* that the ecosystem functions provide. In fact, it is the main purpose of this paper to show that stated preference techniques can be successfully applied to the valuation of ecosystem functions if they are translated into ecosystem services. In line with the most recent recommendations by Heal et al. (2005, p. 121), this effort can be viewed as an extension of insights of the general debate on the economic valuation of ecosystem functions where it is stressed that the economic value of ecosystem functions should be judged by the value of the ecosystem service flows they provide (Freeman, 1998).

In this paper, we first outline the unfamiliarity critique at stated preference methods, and sketch corresponding problems of alternative valuation approaches (Section 2). In Section 3, we propose to regard the difference between ecosystem functions and ecosystem services as an epistemological difference, which results in the proposal for the ecosystem service approach. Based on these ideas, Sections 4 and 5 present and discuss a case study from Central Sulawesi, Indonesia, focusing on the valuation of hydrological ecosystem functioning via a central ecosystem service it provides.

2. Valuing ecosystem functions — the challenges

Unfamiliarity with an environmental good can result in numerous biases that may distort valuation results. In particular, it can result in

- information bias: The less respondents are familiar with the good to be valued, the more their response will be affected by inaccurate, imprecise or even spurious information.
- methodological misspecification bias: Even if the information provided is precise and accurate, there is a risk that respondents do not understand the presented information as intended. Other things being equal, it must be expected that any distortion is the more severe the less familiar the good is.

It is undisputed that these biases have to be taken into account with high priority in the design of stated preference survey instruments (Bateman et al., 2002, p. 76, p. 81, pp. 119ff). For ecosystem functions, unfamiliarity is likely a major problem

[5] Restricting this paper to a discussion of unfamiliarity effects should not be construed as a denial of further issues that continue to be a matter of intensive debate (e.g., Diamond and Hausman, 1994; Sugden, 2005).

as non-expert respondents usually have very restricted knowledge of ecosystem functioning. For ecosystem functions that fall into the indirect use category – including regulation functions of the water cycle –, the suitability of stated preference techniques was explicitly challenged on misspecification bias grounds (Nunes and Bergh, 2001. p. 205). Stated preference methods will fail to "value categories that the general public is not informed about nor has experience with", creating a problematic situation for the valuation of changes in these goods that are "far removed from human perceptions" (Nunes and Bergh, 2001, p. 208, p. 217). Because of the complexity of the ecological interactions that give rise to ecosystem functions, the misspecification problems are unlikely to be remedied by improved design of the survey instrument, Nunes and Bergh claim. Thus, the case against the direct valuation of most ecosystem functions of the indirect use value type by stated preference techniques appears conclusive.

Revealed preference valuation methods provide an alternative valuation approach. These methods rely directly or indirectly on market data. Reliable market data are rare and/or difficult to interpret for many ecosystem functions, however. Thus, it can be expected that revealed preference methods feature their own set of restrictions and imperfections. For example, travel cost and hedonic pricing techniques can only be applied to ecosystem support functions related to tourism and housing. For most ecosystem functions, this relation is absent. For replacement cost calculations, Bockstael et al. (2000) require compliance with three seldom met conditions:

(i) the replacement system provides functions that are quantitatively and qualitatively equivalent to the original ecosystem functions,
(ii) the investigated replacement system is the least cost-option of all potential replacement systems, and
(iii) aggregate willingness-to-pay for the replacement actually exceeds the cost for the replacement in face of the loss of the original ecosystem functions.

A parallel line of reasoning holds for averting costs. The averting measure must be effective in all regards, the least cost-option must be identified, and its costs may not exceed aggregate WTP. According to the review by Heal et al. (2005, p. 191), replacement cost and avoided cost analysis are justified under restrictive conditions only. Production function approaches suffer from the fact that the opportunity costs of switching to the second-best production technology must be known. This can be difficult to achieve if complex substitution and adaptation processes occur, or if policy or production impacts are strongly non-linear. Combined with complex ecological models, considerable progress has been achieved here (Heal et al., 2005, pp. 113ff). However, many functional ecosystem benefits include non-marketed, public option value or insurance value components (Barkmann and Marggraf, 2004). Production functions approaches do structurally not account for these value components.

Although we cannot provide an extensive review of currently available methods for the economic valuation of ecosystem functions and/or services here, we conclude that each method comes with a specific profile of advantages and disadvantages. In practical terms, this suggests that no single approach should be dismissed prematurely as non-applicable — including stated preference techniques.

3. Making sense from ecosystem functions via an ecosystem service approach

3.1. *The background problem*

Because of the multitude of different meanings of the function concept among different sciences, 'ecosystem function' can be interpreted in several, at times conflicting ways (de Groot et al., 2002). The most important conflict exists between a descriptive interpretation and a normative interpretation (cf. Whigham, 1996). In a descriptive interpretation, ecosystem function merely relates to some ecosystem phenomenon that can be thought of as contributing to something else. Accepting the scientific fact of such a contribution does not entail any value judgment. The complication arises, however, that many phenomena to which ecosystem states, processes or structures contribute can have some – sometimes essential – importance for human well-being. Consequently, there exists a normative interpretation, in which an ecosystem function is regarded as an environmental value, as a source of ecosystem benefits in social and/or economic terms. Both interpretations are, as such, legitimate and ultimately a matter of terminological taste. Problems can easily arise, however, in form of a serious category error. This happens if the existence of a decision-relevant normative ecosystem function is inferred directly from the existence of a related ecosystem function in descriptive terms.

Where ecosystem function is explicitly defined in the recent literature, a descriptive interpretation dominates (e.g., Whigham, 1996; Costanza et al., 1997; Freeman, 1998; Heal et al., 2005). If a descriptive notion is favored, a decisively normative notion of ecosystem services suggests itself. In a proposal for a systematic description and valuation of the functions, goods and services of ecological systems, de Groot et al. (2002) "reconceptualise" or "translate" descriptive ecosystem functions into the normative concepts of ecosystem goods and services if and only if human needs or values are affected. Along these lines, several valuation approaches focus on ecosystem services. The Millennium Ecosystem Assessment (MA; Alcamo et al., 2003), for example, concentrates nearly exclusively on ecosystem services (also see, Toman, 1996; Heal et al., 2005).

In the MA, ecosystem services are defined as

> "the benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other non-material benefits." (Box 1, Key Definitions, Alcamo et al., 2003, p. 3)

In line with MA terminology, we regard ecosystem services as immaterial economic goods provided by ecological systems including their elements, structures, processes, states, dynamics etc. If an ecosystem provides a material good, such as water used for irrigation, the process of provisioning the material good is an ecosystem service.

In their ecosystem and biodiversity valuation critique, Nunes and Bergh (2001) develop a detailed classification of biodiversity values. They do not differentiate systematically, though, between functions and services. For example, "functional diversity" (Nunes and Bergh, 2001, p. 204) encompasses "primary ecosystem processes" (photosynthesis and biogeochemical cycling) – ecosystem functions in a descriptive sense. Functional diversity also encompasses "ecosystem life support functions", e.g., the regulation of water and carbon cycles – as regularly included in lists of ecosystem services (e.g., Heal et al., 2005, pp. 80–83). If combined with the unfamiliarity critique, the lack of a clear differentiation may foster the impression that stated preference methods are principally unsuitable for the valuation of ecosystem functions.

3.2. *The ecosystem service approach*

Lay respondents are utterly unfamiliar with scientific descriptions of the processes, structures and states that make up ecosystem functions. Thus, it is virtually impossible to sample meaningful economic preference statements for ecosystem functions described by basic science models (Fig. 1a). As an example, we included an exceedence time graph in the following case study section that represents a central hydrological ecosystem function in a small Central Sulawesi catchment (Fig. 3). Although simple to interpret for the scientifically trained analyst, very few stated preference respondents will muster the patience to learn enough about hydrology during the valuation interview to do so.

Even worse, without additional engineering and agricultural data, even experts can only make an educated guess how changes in the exceedence time graph may correlate with improved water availability for wet rice cultivation in the dry season in a heterogeneous multi-catchment project area. Mismatches between the ecosystem function output of basic science models and the requirements for economic analysis are rather common (see example by Freeman, 1998, p. 249f).

The situation improves if we employ an ecosystem service approach to environmental valuation (Fig. 1b). Precursors of the ideas presented here are provided by Carson et al. (1999), Pattanayak and Kramer (2001), and Bennett (2002). Using the ecosystem service approach, we invest a certain measure of normativity that allows us to differentiate between ecosystem states, structures and processes that do in fact contribute to human production and consumption, and those that – to the best of our knowledge and with reference to the valuation task at hand – do not. With this differentiation in mind, engineering models can be constructed that are concerned with socially relevant ecosystem services (ecosystem services I). The unfamiliarity of respondents with model outputs decreases. For communication with lay stakeholders, however, model outputs are often still not suitable because engineering models themselves are usually expert models. An additional step is required to translate ecosystem services I into the language of the interests, concepts and perceptions of lay respondents (transdisciplinary model; ecosystem services II). This step requires substantial qualitative social science research into the subjective, pre-theoretic patterns of perception and valuation of the natural phenomena at stake (Barkmann et al., 2005a).

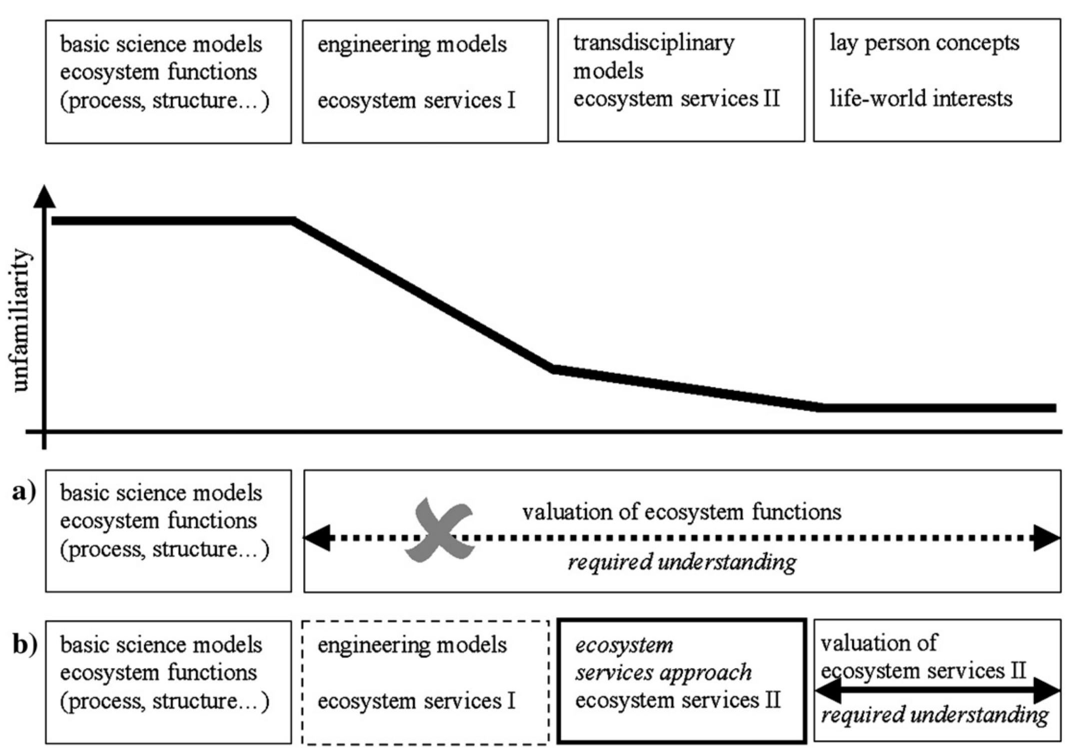

Fig. 1 – Valuation of ecosystem functions versus valuation of ecosystem services: advantages and disadvantages in the face of unfamiliarity of lay respondents with detailed scientific knowledge; (a) stated preference valuation of ecosystem functions; (b) stated preference valuation of ecosystem services II (oringinal graph).

Thorough qualitative investigations of the pre-theoretic concepts, beliefs and values that non-expert respondents are likely to bring to the valuation task are a standard requirement for any *lege artis* empirical stated preference study (e.g., Arrow et al., 1993; Bateman et al., 2002). The ecosystem service approach differs from standard procedures in terms of an additional restriction on commodity definition and the ensuing quantitative description of CE attribute levels or CV alternatives. First, the good to be valued must be described as an ecosystem service, i.e. in terms of a benefit obtained from ecological systems that respondents actually care for. This requirement excludes those functional 'commodities' (i.e., ecosystem functions) from valuation for which the analyst is unable to clearly identify such benefits. In consequence, the quantitative description of stated preference attribute/alternatives levels must be given in benefit terms. A quantification in bio-physical terms will usually not be sufficient. If valuations are relative to some status quo situation, the benefits associated with the status quo must also be spelled out.

4. Case study: valuating a hydrological ecosystem service in rural Indonesia

Being aware of the criticism expressed at the application of stated preference methods to hydrological ecosystem functions, a valuation study was designed that aimed at minimizing information and misspecification biases by a careful differentiation between ecosystem structures and processes, and the ecosystem services they generate (Barkmann and Marggraf, 2003, p. 248). Although the presentation below focuses on hydrologic aspects, the case study itself deals with four ecosystem services of which only one is related to ecosystem hydrology (see Table 1). This broader scope is mainly due to the institutional framework of the Indonesian-German collaborative research project "Stability of Rainforest Margins in Indonesia" (www.storma.de). Here, not only hydrological but also micro-climatical, pedological, zoological and botanical impacts of forest conversion and agroforestry intensification are investigated (Steffan-Dewenter et al., 2007).

With only one choice experiment attribute devoted to hydrological ecosystem services, the case study cannot provide preference estimations for the full range of services that depend on local hydrologic ecosystem functioning. Instead, the case study (i) illustrates how the ecosystem service approach can be implemented, and (ii) tests its applicability with a focus on an ecosystem function for which the applicability of stated preference methods has explicitly been challenged.

4.1. *Description of the research area*

The economic valuation exercise took place in the area of the Lore Lindu National Park in Central Sulawesi, Indonesia. The project area of about 7220 km^2 is inhabited by ~137,000 people, mostly agricultural smallholders (data from 2001; Maertens et al., 2004). The area is part of the globally important Wallacea biodiversity hotspot.

The research area is characterized by substantial environmental heterogeneity. The altitude ranges from just above sea level up to 2500 m, and rainfall varies from 500 mm to 2500 mm/year (Maertens, 2003). For example, some villages are located in the intensively used Palu River Valley, close to the provincial capital and well connected to national markets. The intensity of agricultural land use – including more intensive cacao agroforestry devoid of a dense shading tree canopy (Steffan-Dewenter et al., 2007) – decreases via villages in the northeast of Lore Lindu National Park (Palolo Valley) and the immediate east and west (Napu and Kulawi valleys) still accessible by paved roads to upland villages in the remote west not even accessible by motor bike. These differences coincide partly with the local degree of rattan exploitation, and with an altitude gradient that severely restricts cacao production, e.g., in the mountainous west. Furthermore, the local availability of irrigation water differs markedly depending on rainfall differences (semi arid climate in the Palu valley) and on the type of irrigation systems available. While several villages in the Palu River Valley are connected to the (technical) Gumbasa Watershed Irrigation System, all other villages use semi-technical or traditional irrigation systems fed from local water sources.

In one of the research villages, Nopu (Palolo Valley), Keil et al. (2003) investigated the effects of forest conversion on water availability and local water use. For approximately ten years, smallholders have cultivated cacao in the floodplains of the catchment, where the village is located. Between 650 and 950 m above sea level, the tropical forest is rapidly being replaced by

Table 1 – Attributes and attribute levels of the Central Sulawesi choice experiment

	Attribute	Levels	Ecosystem service category	Value type (TEV)
Rattan	Availability of rattan (*Calamus* spp.) expressed in distance from village	5, 10, 15, 20 [km]	Provisioning service	Direct use/option value
Water	Availability of irrigation water for wet rice cultivation	0, 1, 2, 3 [number of months with water shortage]	Regulating service	Indirect use value
Cocoa	Preponderance of cocoa plantations differing along a shade tree gradient	5, 35, 65, 95 [% under shade]	Regulating services	Indirect use/option value
Anoa	Population size of the endemic dwarf buffalo anoa (*Bubalus depressicornis/quarlesi*)	10, 180, 350[§], 520 [No. of animals]	Cultural service	(Predominantly) Existence value
Cost	Extra taxes or donation to village fund	0, 18, 36, 54, 72 [1000 IDR/yr[$]]	–	–

[§] present state; [$] split sampled attribute: cards with a monthly payment scheme display 1/12 of the yearly amounts; 1 € ~ 11,500 IDR (winter/spring 2005).

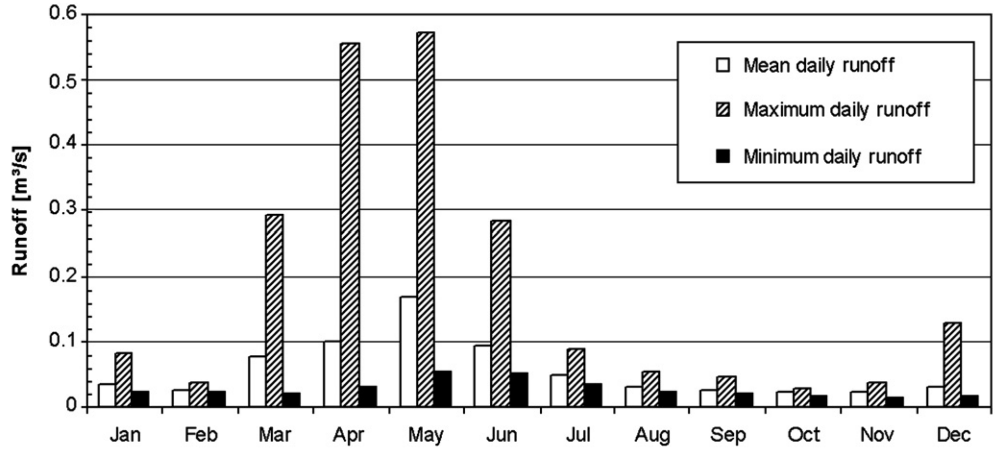

Fig. 2 – Monthly minimum, maximum and mean daily runoff of Nopu catchment, Central Sulawesi, in 2002 (Keil et al., 2003).

patches of slash-and-burn agriculture, pasture, secondary forest, and cacao agroforestry. To varying degrees, these processes characterise the entire project area around Lore Lindu National Park.

Since September 2001 the Nopu catchment is instrumented with water level recorders, hydrologic parameter sensors, rain gauges, and meteorological stations. As an example of scientific key findings on hydrologic ecosystem functions, we present data from 2002 that were already available when the economic valuation study started in July 2003. The river discharge distribution (minimum daily runoff) shows a seasonal pattern with a peak in June and a low in November reflecting a spring/early summer peak in precipitation (maximum daily runoff (Fig. 2). Between months with a peak in precipitation and months with a peak in minimum discharge, there is a time lag of about two months.

One of the most appropriate scientific descriptions of the hydraulic ecosystem functions with relevance to water supply are exceedence time graphs (Fig. 3). Exceedence time graphs indicate the number of days of a year in which river discharge exceeds the values indicated by the ordinate of the graph. Daily discharge values of Nopu river below 0.05 m^3 indicate low water availability for about 2/3 of the year.

4.2. *Design and application of the choice experiment*

4.2.1. *Design of the attributes*

Blamey et al. (1997) differentiate between demand-relevance and policy-relevance of goods included in stated preference studies. Demand-driven goods are expected to show up spontaneously in focus groups or semi-structured interviews because respondents perceive the natural resource management issue as pressing. If the issue is not mentioned spontaneously, there is a higher risk that respondents are not familiar with the issue. Consequently, information and misspecification biases are a more serious threat to validity than for demand-driven attributes.

Information gathered from 120 households in 12 villages of the project region did already suggest in 2002 that there is a high awareness of drought and flooding related problems (Birner and Mappatoba, 2002). Keil et al. (2003) specifically addressed perceived water availability in a one-day Participatory Rural Appraisal (PRA) held in Nopu in May 2002. Participants were asked to assess monthly rainfall and local water availability for a 'normal' year (Fig. 4). The subjective assessment of water availability follows perceived rainfall patterns with a delay of up to two months. Although the

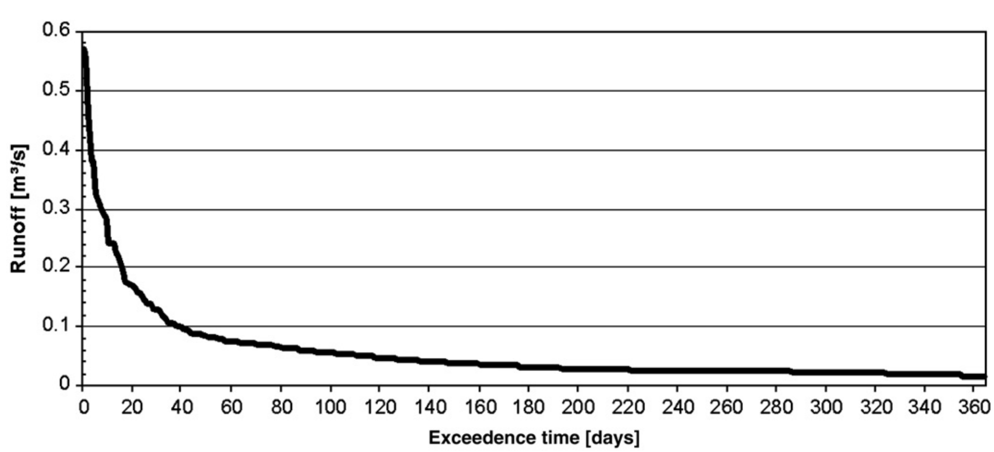

Fig. 3 – Exceedence time graph 2002, Nopu catchment, Central Sulawesi (Keil et al., 2003).

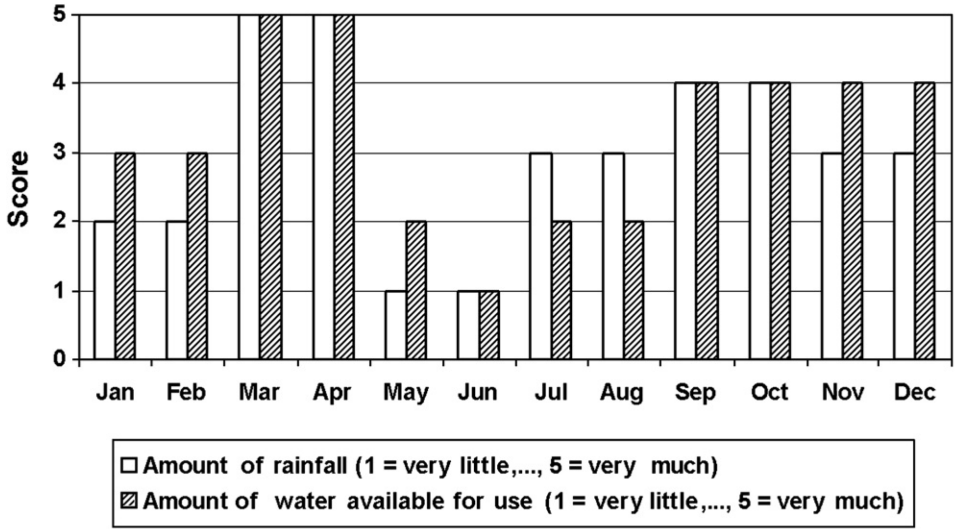

Fig. 4 – Average rainfall distribution and water availability in Nopu, Central Sulawesi, for a 'normal' year (PRA results, modified from Keil et al., 2003).

seasonal distribution of precipitation and river discharge in 2002 differed from that of a 'normal' year, the gap between peaks in precipitation and minimum river discharge is mirrored in the perceptions of PRA participants. Additional focus group and semi-structured in-depth interviews were conducted in several villages of the study region in 2003. Problems with water regulation and water shortage figured prominently as resource management issues in a forest-related context, i.e. at an ecosystem level (Fig. 5). Consequently, we encountered favourable conditions for the identification of demand-driven hydrological ecosystem functions.

In a study exclusively focusing on the valuation of hydrological ecosystem services, water availability for irrigation as well as the provision of potable water or flood protection could have been included as separate choice experiment attributes. Because of pragmatic reasons, only one CE attribute could be devoted to the valuation of hydrological ecosystem services, however (for a list of attributes, see Table 1). We decided to focus on water availability during the dry season for wet rice farming because most water in the project area is used for irrigation of wet rice. The attribute levels are quantified by different combinations of months with enough, hardly enough, and not enough water. By focusing on this aspect, we deliberately ignore other hydrologic ecosystem services also deserving close investigation. It was considered to use a more general attribute on water availability that includes water for wet rice irrigation as well as water for human consumption. However, as benefits and water quality requirements differ (e.g., turbidity, human pathogens), this would have resulted in a more comprehensive but also substantially more complex attribute. In sum, an attribute on water availability for wet rice farming during the dry season appeared to be a suitable – albeit partial – translation of the

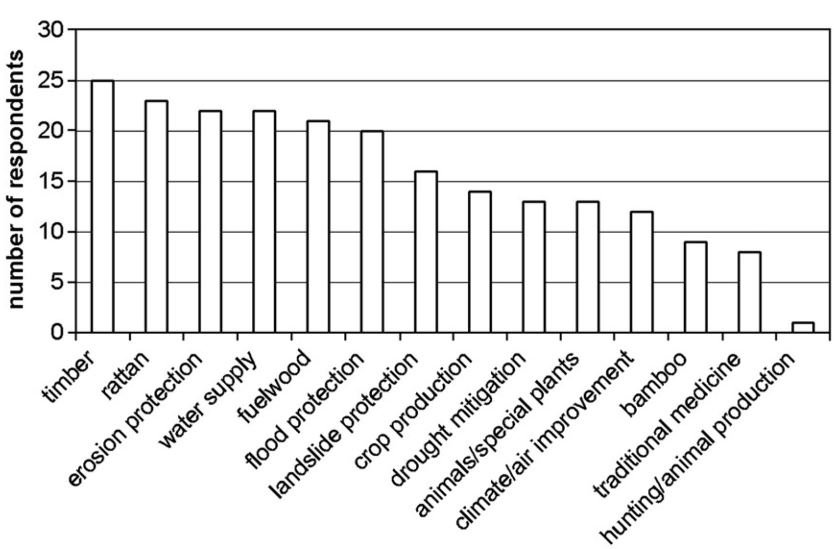

Fig. 5 – Perceived forest benefits: results of 26 pre-study interviews conducted in several villages of the Lore Lindu area in 2003 (original data).

scientific description of hydrological processes into a demand-driven ecosystem service.

In addition to the water availability attribute, we selected the population size of an endemic local species, the dwarf buffalo *Bubalus depressicornis/B. quarlesi* ('anoa'), the amount of shade in local cacao plantations, and the availability of rattan as additional attributes (Table 1). In exact analogy to the water attribute, the rattan attribute was operationalised via the ecosystem service approach (distance to next commercially exploitable rattan site in km). Due to conservation policy interest in the local economic perception of varying cacao shading levels by Central Sulawesi small holders (cf. Steffan-Dewenter et al., 2007), the cacao attribute was not operationalised via the ecosystem service attribute but presented in structural terms. For the anoa attribute, the ecosystem service approach was inappropriate as the anoa population is not valued because of any ecosystem function in the sense of an indirect use value but mainly for existence value reasons. Although hunting anoa is illegal, we cannot rule out that some respondents also attributed a direct use value to the anoa population. In any case, this direct use service can be considered highly familiar to respondents, and not in need of a specific translation into respondent 'benefit language'.

Although most respondents are involved in agricultural subsistence production, pre-studies had shown that virtually all households are also involved in money-based economic transactions, either via the purchase of agricultural inputs, or via selling raw materials for internationally traded commodities cacao and rattan. Thus, we opted to use the monetary standard version of the payment vehicle. Because of the severe poverty of many respondents, we restricted maximum monetary levels – accepting an underestimation of WTP by ignoring potentially higher WTP of a low percentage of rather well-off respondents. The 'cost' attribute was double split-sampled. One half of respondents were confronted with a rise in "house-and-land" tax (*Pajak Bumi Bangunan*), the other half with a donation to a village fund (*Iuran dana pembangunan desa*) affecting every household of the research region. Both payment vehicles are familiar and widely accepted within the region. The second split sample involved monthly versus yearly payments. Results of the split sample analysis will not be reported here.

To alleviate the cognitive burden of the choice task, we used visualizations for the attributes and attribute levels crafted by a local artist. The choice task was framed as a selection between different versions of a village development program to which a financial contribution was required if implemented.

4.2.2. *Administration and analysis of the choice experiment*

The choice experiment (CE) instrument was pre-tested and piloted ($n=96$) in 2004. The main survey was conducted from December 2004 to March 2005 as a random face-to-face survey in 12 of the about 120 villages in the project area ($n=301$). The interviews were conducted by six well-trained local enumerators. Thirteen respondents consistently chose the status quo without quoting lack of money or low attractiveness of the offered alternatives as reasons. These respondents were classified as not responding to the CE task and omitted from parameter estimation (Adamowicz et al., 1998, p. 68). Quantitative analysis is performed on a subset of 249 households for which an extensive set of socio-demographic variables is available (for descriptive statistics, see Table 2).

Because of the variability of ecological conditions across the project region, the choice sets presented to a respondent included a self-explicated status quo (Bennett and Blamey, 2001, p. 138; for details of application, see Glenk et al., 2006a). Because of these pronounced differences in the project region with regard to the water, the rattan and the cacao attribute, respondents were asked which of the attribute levels (see

Table 2 – Socio-economic and attitudinal description of the sample of Central Sulawesi respondents/respondent households

Variable	Type	Coding	Scale construction	Mean	S.D.
Respondent age	SD		n.a.	45.8	14.0
Respondent education [years][§]	SD		n.a.	7.6	3.2
Relative poverty	SD	see Zeller et al. (2003)		0.02	0.81
Rattan collector[$]	SD	0;1	n.a.	0.12	0.33
Wet rice involvement[$]	SD	0;1	n.a.	0.60	0.49
Cacao farmer[$]	SD	0;1	n.a.	0.76	0.43
Perception of % shading in local cacao plots	SSQ	respondents choose from cacao shading attribute levels (5%; 35%; 65%; 95%)		42.8	18.2
Rattan value	PMT	1;…; 4	mean of two items	1.55	1.05
Probability of rattan loss	PMT	1;…; 4	mean of two items	2.26	0.97
Water value	PMT	1;…; 4	one item	3.10	1.34
Severity of forest loss	PMT	1;…; 4	one item	3.25	0.98
Cacao shade value	PMT	1;…; 4	one item	1.51	0.96
Probability of cacao problems	PMT	1;…; 4	one item	3.00	1.06
Severity of anoa extinction	PMT	1;…; 4	one item	1.29	0.59
Government response efficacy	PMT	1;…; 5	mean of two items	4.08	0.47
Respondent comprehension rating	RCR	1;…; 5	n.a.	3.12	0.85

SD: socio-demographic variable; SSQ: self-explicated status quo; PMT: Protection Motivation Theory (Rogers and Prentice-Dunn 1997); RCR: rating of respondent comprehension performed by enumerators immediately after the choice experiment; [§] of the mean, 6 years are primary school; [$] mean indicates share of households involved in respective activities; $n=249$.

Table 3 – Valuation of functional ecosystem services in Central Sulawesi (Indonesia)

Variable	(a) Base model	(b) Direct utilisation	(c) PMT and relative poverty	(d) PMT and respondent comprehension
Rattan availability	−0.0457***	−0.0334***	−0.0707**	−0.0267
Water for irrigation in the dry season	−0.9771***	−0.5742***	−0.0102	0.4222
Cacao Shade (linear)	0.0173*	0.0288***	0.0447***	0.0458***
Cacao Shade (quad.)	−0.0003***	−0.0003***	−0.0003***	−0.0003***
Anoa Population Size	0.0013***	0.0014***	−0.0007	0.0001
Cost	−0.0247***	−0.0256***	0.0174	−0.0029
Cost*Response Efficacy			−0.0107**	−0.0107**
ASC (non-status quo)	1.4081***	1.4106***	1.5461***	2.1543***
ASC*RCR	−0.2735**	−0.3021***	−0.3261***	−0.5322**
Rattan*rat. collector		−0.1101***		
Water*involvement		−0.7305***		
Cacao*cacao farmer		−0.0130**		
Rattan*rattan value			−0.0286**	−0.0281**
Rattan*Probability			0.0284***	0.0279**
Water*water value			−0.2181***	−0.2077***
Water*severity of forest loss			−0.1033*	−0.0925(T)
Cacao*value			−0.0043*	−0.0041*
Cacao*probability			−0.0061***	−0.0060**
Cacao*relat. poverty			−0.0042(T)	−0.0039
Anoa*severity			0.0017**	0.0016**
Rattan*RCR				−0.0146
Water*RCR				−0.1658*
Cacao*RCR				−0.0006
Anoa*RCR				−0.0002
Cost*RCR				0.0048
Log-likelihood	−728.2	−690.0	−680.6	−678.1
P(Chi^2); DF	<0.0001; 9	<0.0001; 12	<0.0001; 18	<0.0001; 23
Inclusive value (IV)#	0.8576	0.8695	0.8563*	0.7789*
Adj. ρ^2 (Pseudo-R^2)§	0.273	0.3117	0.319	0.3198

(T) tendency at $p<0.1$; cost coefficients for 1000 IDR/yr; & IV of non-degenerated branch; degenerated branch set to 1.0; § Pseudo-R^2 values in reference to a *constants only* model — values between 0.3 and 0.4 correspond to R^2 values of 0.6 to 0.8 value in ordinary least squares regression (Hensher et al., 2005, p. 338); # all IV statistics are highly significantly different from 0, asterisks denotes difference from 1; RCR: respondent comprehension rating; DF: degrees of freedom; Nested Logit models based on 249 respondents with 4 choices each: $n=996$ observations.

Table 1) most closely described the local situation of their village.[6] For each respondent, these levels were individually affixed to the status quo choice card. For the anoa population of the research region, an expert estimate of 300 animals was uniformly used as the status quo level.

For all attributes, linear utility functions were assumed. For cacao shading, an additional quadratic term was included in order to allow for a shading optimum at intermediate shade levels. Preliminary analyses indicated the risk of violations of the independence from irrelevant alternatives (IIA) condition necessary for the application of multi-nomial logit analysis. Because Nested Logit (NL) analysis does not rely on the IIA assumption, an eligible NL tree structure was identified, and the corresponding models estimated with NLOGIT 3.0. The inclusive value was set to 1.0 for the degenerated branch, and the model initiated with starting values obtained from a non-nested NL model (Hensher et al., 2005, pp. 530ff). All scale parameters were normalized at the lowest level (RU1). Willingness-to-pay calculations are based on extrapolations from mean marginal WTP values. Individual self-explicated status quo levels were accounted for in calculation of WTP values of the cacao attribute: (i) For respondents perceiving 5% shading, scenario change was restricted to −5%; (ii) mean scenario WTP/household was calculated separately for each group of respondents with differing status quo self-explications; (iii) average scenario WTP per household is the weighed mean of the WTP values across the groups. For the final aggregation of WTP across the households of the project region, we regarded 13 non-responding participants as filing protest statements indicating zero WTP.

4.2.3. *Validity tests and influence of respondent comprehension*

Four NL models are calculated that show basic results (Table 3, model a), and allow for the evaluation of a number of validity

[6] Some people were not confident or able to state a perceived level for some attributes. In such cases, we drew on information from meetings with village officials and, especially for rattan, a range of farmers more intensively involved in rattan extraction. There were cases where substantial deviations were reported (e.g., no water scarcity vs. 3 months of scarcity). Invariably, these cases were due to intra-village heterogeneity, e.g. regarding the performance of the local irrigation system. This observation further supports the case against an identical status quo — even at the village level.

issues (Table 3, models b–d). The validity tests address three potential challenges to the results of the case study.

First, it could be claimed that we do not measure the value of functional *indirect* use benefits at the ecosystem level, but the value of *direct* use benefits at the level of the individual farm household that, e.g., directly uses irrigation water. Although the actual irrigation water needs to be "produced" from the water available at the ecosystem level by the construction and maintenance of irrigation systems and, thus, cannot be used directly for irrigation, we decided to conduct a simple test if preferences are exclusively linked to the economic activities of the individual households (model b). We do so by including an interaction term with a dummy variable indicating if a household is involved in wet rice farming, extracts rattan, or owns a cacao plot.

The second challenge refers to the general validity of CE results in face of the hypothetical character of stated preference methods. In model (c), we use attitudinal items that operationalise risk and coping appraisal variables from Protection Motivation Theory (PMT, Rogers and Prentice-Dunn, 1997). PMT is a social-psychological action theory that has been used successfully for the explanation of economic preferences calculated from stated preference data (Menzel, 2003; Barkmann et al., 2005b). An additional set of PMT items was sampled on forest-related attitudes. Six items were sampled that refer to respondent attitudes towards the government's ability to use taxpayer money successfully and responsibly for village development. For quantitative analysis, we use a two-item subscale singled out by factor analysis that focuses on attitudes beyond village government (for details, see Table 2).

A standard procedure to test for validity in stated preference studies relies on the expected positive influence of income on WTP. Under the semi-subsistence conditions prevalent in the project area, we tested for more broadly defined wealth effects on preferences using a relative poverty index (Glenk et al., 2006b; Zeller et al., 2003).

The third challenge refers to respondent unfamiliarity with the ecosystem services sampled. Unfamiliarity may result in poor comprehension of the subject matter of the choice experiment. Enumerators rated respondent comprehension directly after the choice experiment on a scale from 1 (= insufficient understanding) to 5 (= very good understanding). The average comprehension rating was 3.12 with only four respondents in the 'insufficient' category (Table 2). With this indicator of respondent comprehension, we test three hypotheses:

1. Respondent comprehension influences preferences for ecosystem services. Influences are tested by constructing interaction terms between attributes and comprehension rating.
2. Inclusion of the interaction terms does not result in materially differing preference estimates.
   Both hypotheses are tested in model (d), into which the interaction terms are introduced in addition to the interaction terms from model (c).
3. Lower comprehension as a result of unfamiliarity results in a less thorough cognitive integration of the information given in the choice sets. We test this hypothesis
   a. by observing the sign of a dummy variable representing the non-status quo alternatives ('ASC'). If respondents feel ill-informed with regard to the costly alternatives to the status quo, one would expect a negative sign of the ASC (Kontoleon and Yabe, 2004);
   b. by constructing an interaction term between the respondent comprehension rating and the ASC. If hypothesis 3 is true, the interaction term should become a significant predictor of choice, and its sign should be opposite to the sign of the ASC.

The ASC dummy and the interaction term with the ASC dummy are included in all four models. Details on the econometric treatment of interaction terms in utility and WTP calculation can be found in Barkmann et al. (2007).

4.3. Results

4.3.1. Base model, validity tests

All NL models are highly significant ($P \leq 0.0001$; Table 3). The base model (a) displays the expected signs for the attribute terms: disutility for longer distance to the next rattan site, disutility of decline in water availability, positive utility for bigger anoa populations, disutility of higher costs. For shading in cacao plots, we find positive utility for more shade along with a negative sign of the quadratic term indicating an unimodal response to shade. The attribute on preferences for the hydrological ecosystem service availability of irrigation water in the dry season is highly significant ($P<0.0001$).

Model (b) reveals that for rattan, water and cacao shade, the direct utilization terms become highly significant; the non-interacted attributes indicating non-production preferences are also highly significant. Taking the fraction of households into account that are involved in rattan extraction and wet rice farming (Table 3), a comparison of the rattan-and water-related coefficients suggests that about 71% of the preferences for improved rattan availability are attributable to non-production benefits of the individual household; for improved availability of irrigation water, about 57% are attributable to non-production benefits.

Comparing model (c) with the base model (a), inclusion of significant interaction terms with PMT variables and with relative poverty improves adjusted Pseudo-R^2 by more than four percentage points to a very reasonable value of 31.9%. Log Likelihood ratio-tests show that model specification (c) improves model fit compared to models (a) and (b) ($P<0.0001$; $P=0.027$). Significant interaction terms are found for all attributes. Two interactions with the water attribute are significant, water value ($P<0.0001$) and severity of forest loss ($P=0.037$). The more respondents agree on average with the statement that they "very much need water for irrigation in the dry season" (water value) the higher is the disutility of more months with low water availability. Likewise, a more concerned attitude towards loss of "trees on the hillsides" correlates with higher preferences for water availability. Both interaction terms explain variation in choice behavior as expected. In fact, they explain the variation so well that the non-interacted water attribute becomes statistically and materially insignificant. The cost attribute displays the same pattern with a highly significant influence of the perceived response efficacy of government ($P=0.0039$).

Table 4 – Impact of the inclusion of the respondent comprehension rating on disutility for one additional month with water scarcity for an "average" household

Variable	Model coefficients		Mean of interacted variable	Interacted coefficients	
	Model (c)	Model (d)		Model (c)	Model (d)
Water for irrigation in the dry season	−0.0102	0.4222		−0.0102	0.4222
Water*water value	−0.2181	−0.2077	3.10	−0.6770	−0.6449
Water*severity of forest loss	−0.1033	−0.0925	3.25	−0.3360	−0.3008
Water*RCR		−0.1658	3.12		−0.5167
Sum Water				−1.0232	−1.0403
Cost$	0.0174	−0.0029		0.0174	0.0029
Cost*response efficacy	−0.0107	−0.0107	4.08	−0.0436	−0.0438
Cost*RCR		0.0048	3.12		0.0149
Sum cost				−0.0262	−0.0261
Marginal WTP§ [IDR/yr]				−38,979	−39,927

RCR: Respondent Comprehension Rating; $ all cost coefficients for 1000 IDR; § marginal WTP = −(Sum Water/Sum Cost)*1000.

Under model specifications (c) and (d), none of the theoretically expected linear influences of relative poverty on preferences could be detected at standard significance levels. Only the interaction with the cacao attribute displays a tendency ($P=0.075$) in model (c); only the interactions with cacao are retained in the final models.

Model (d) addresses respondent comprehension. Only the interaction of the comprehension rating and the water attribute influences choices ($P=0.019$). Model (d) predicts that a better comprehension rating results in higher disutility of water scarcity, i.e., better understanding respondents have stronger preferences for the ecosystem service water availability than respondents with a lower comprehension rating. The next best interaction term is the interaction term with rattan ($P=0.194$). Table 4 details that the inclusion of the respondent comprehension rating results in virtually identical coefficients of models (c) and (d) when summed up across all relevant terms (see hypothesis 2). Accordingly, the two marginal willingness-to-pay estimations for one month of reduced water scarcity around 39,000–40,000 IDR/yr hardly differ.

If we stipulate counter-factually that respondents had a mean comprehension rating of 4 instead of just above 3, model (d) predicts that the sum of the water coefficients would increase by 14.1% (−0.146). In this calculation, we keep the coefficient of the interaction cost attribute-comprehension rating constant because it is clearly not significant ($P=0.215$).

In addition to the analysis of respondent comprehension at the level of individual attributes, we tested if potential unfamiliarity effects influenced choice between the status quo option and the village development program — i.e., the non-status quo alternatives. As hypothesized, the interaction term of the ASC-dummy representing the non-status quo alternatives consistently influenced choices (Table 3, models a–d). The significance and positive sign of the ASC itself indicate that respondents favor the proposed village development program beyond the actual ecosystem service change presented in the scenarios. This effect is partially counterbalanced by the interaction term between ASC and respondent comprehension rating because its sign is opposite to the sign of the ASC coefficient. For respondents with average comprehension, the effect is reduced by nearly one half. Thus, hypothesis 3 that claims that lower comprehension results in worse cognitive integration of the information actually offered in the choice experiment could not be rejected. The sign of the ASC is not consistent, however, with high levels of subjective uncertainty caused by unfamiliarity (test 3a). In this case, the uncertainty should have lead to a rejection of the offered choices in favor of the status quo beyond the information given. This would have been expressed by a negative sign of the ASC.

4.3.2. *Willingness-to-pay calculation*

For an estimation of marginal WTP values, we return to model (c), which fits the data no worse than model (d) but which contains only one insignificant interaction term (cacao*relative poverty; Log Likelihood ratio-test of model fit improvement: $P=0.834$). Marginal mean values range from about 400 IDR/household/yr for cacao shading to about 39,000 IDR for improved water availability (Table 5). Additionally, Table 5 presents a

Table 5 – Marginal WTP and exemplary scenario calculation based on model (c) statistics

	Rattan	Water	Cacao	Anoa
Marginal Unit	harvesting location 1 km closer	1 month less with water scarcity	1% less shading in plantations of village	1 individual more
Marginal mean WTP/household [IDR/yr]	1,936	38,979	408$	49
Scenario change	−10 km	−1 month	−20%§	+150
Mean scenario WTP/household	19,362	38,979	4,438#	7,386
Aggregated scenario WTP/project area [10^6 IDR/yr]	431.8	869.2	98.8	164.7

$ accounting for self-explicated status quo (see explanation in Section 4.2.2); § for respondents perceiving 5% shading, scenario change = −5%; # because of non-linear preferences for cacao shading, mean scenario WTP ≠ marginal mean WTP*scenario change; aggregation based on 23,307*(301−13)/301=22,300 households in the project region accounting for 13 protest responses; 1 €~11,500 IDR (winter/spring 2005).

simplified scenario calculation to indicate the magnitude of potential policy implications. The scenario assumes that the next commercially exploitable rattan sites are 10 km closer to the villages, water shortage is reduced by one month, cacao shading is reduced by 20%, and anoa numbers increase by 150. With respect to scenario WTP values, the improvement of hydrological ecosystem services is the most valued change. A methodologically more intricate WTP and scenario analysis with three differing scenarios is provided by Glenk et al. (2006a). Extrapolating household WTP/year to the entire project region of approximately 23,300 households based on the above scenario results in an aggregated annual WTP of about 870 Mio. IDR (~76,000 €) for improved availability of irrigation water, and about 1,560 Mio. IDR (~136,000 €) for an implementation of all improvements.

5. Discussion

5.1. *Unfamiliarity aspects*

The case study documents that unfamiliarity of lay respondents with the scientific details of ecosystem functioning is no principal objection against the application of stated preference techniques for a monetary quantification of 'functional' indirect use values. In particular, we showed that unfamiliarity with technical descriptions of a hydrological ecosystem function represented by the discharge characteristics of a small catchment does not prevent a meaningful economic valuation of a closely related ecosystem service.

Apart from statistical evidence such as the high significance of all calculated NL models and the water availability-related attributes, we base this optimistic view on several observations. At the most fundamental level, water availability is clearly a demand-driven environmental valuation issue of high economic and social relevance in the project region. This is an important factor that helps avoiding information and misspecification biases. The relevance and realism of the entire valuation exercise is mirrored by the fact that only 13 protest responses were filed by 301 respondents.

We investigated if the ecosystem services at hand could be explained completely by motivations for direct resource use at the household level. If so, the need to apply stated preference techniques could be questioned. In particular, for the availability of irrigation water, a substantial share of benefits (57%) could not be attributed directly to wet rice production activities at the household level. Thus, the measured preferences are likely to include non-use and/or indirect use values not captured by standard production economic analysis.

Two variables from Protection Motivation Theory interacted with the water attribute, water value and severity of forest loss, explain variations in choice behavior so well that the water attribute itself became insignificant. Thus, preferences stated in response to the chosen representation of water availability relate well to underlying values and risk perceptions of respondents derived from social-psychological theory.

Preferences for the availability of water and rattan, and for a larger anoa population did not increase linearly with relative poverty. In more detailed analyses to be published elsewhere (Glenk et al., 2006b), preferences for shading in cacao had displayed a (negative) linear correlation with relative poverty. This tendency is also visible here in model (c). As there are indications for a systematically non-linear relationship between relative poverty and preferences for these ecosystem services ('middle class goods'), the results of this validity analysis are inconclusive.

To test for unfamiliarity effects, the comprehension of respondents was rated directly after the CE by local, well-trained enumerators. Only four respondents were rated to have insufficient comprehension. The average comprehension rating was 3.12 with about 80% of respondents in the average to very high comprehension range. Although the comprehension rating influenced the probability with which the village development alternatives were chosen over the "do nothing"-status quo option, the sign of the non-status quo dummy does not give any indication of comprehension problems. With unfamiliarity issues looming, one would have expected a status quo bias (negative sign of the non-status quo ASC) as respondents choose the save haven of the current situation instead of paying for a change in a good troubled by unfamiliarity. Instead, respondents favored the non-status quo alternatives strongly (for a more detailed discussion, see Glenk et al. 2006a).

When interacted with the water attribute, the comprehension rating significantly influenced choices. Potential unfamiliarity effects were stronger for the hydrological ecosystem service than for other ecosystem services, such as availability of rattan or shading in cacao plantations. While the inclusion of the comprehension rating had hardly any effect on WTP estimates, improved familiarity may result in different – in our case higher – WTP estimates. If we attribute comprehension deficits completely to persisting unfamiliarity – which is certainly an exaggeration –, the comprehension rating indicates that unfamiliarity may continue to be a problem for about 20% of respondents. NL model (d) predicts that respondents with a comprehension rating of 4 are expected to value improved water availability by about 14% more than respondents with an average rating. While perceived response efficacy of government activities strongly impacts monetary preferences via an influence on the cost attribute, no evidence was found that unfamiliarity exerts a similar influence.

The comprehension analysis suggests that our implementation of the ecosystem service approach solved the comprehension problems sufficiently when applied to a respondent sample with an average of 7.6±3.2 years of formal education (Table 2). For remaining comprehension problems caused by unfamiliarity, the inclusion of a comprehension rating variable allows for an assessment of the maximum unfamiliarity impact — and a correction of WTP estimates if desired.

The calculated WTP values per household of a few Euro/year appear low. However, WTP for improving water availability by one month equals about 1% of mean cash income of households in the project area (total mean income including non-cash farming income in 2002: ~5.9 Mio IDR; Schwarze and Zeller, 2005). WTP for the implementation of all scenario improvements covers the basic needs of one household member for about half a month (van Rheenen et al., 2004). To our knowledge, the only stated preference study on the valuation of a similar ecosystem service in Indonesia was conducted by Pattanayak and Kramer (2001). They conducted a contingent valuation study on improved drought mitigation services by watershed protection around Ruteng Park (Flores Island). They report a comparable WTP of 2–3 $ per household/year.

Also the WTP value aggregated across the about 23,300 households of 76,000 €/yr for one month of improved water availability appears low. For the conservation of Leuser National Park on Sumatra, van Beukering et al. (2003) calculated annual net benefits for averted losses in water supply alone equivalent to 98 Mio. US$/year. Even considering the fact that Leuser National Park is about three times the size of the Lore Lindu project area and taking vastly differing scenario assumptions into account, differences of about two orders of magnitude remain. These differences are most likely caused by

(i) the restriction of our study to water availability for irrigation (versus drinking water and industrial uses),
(ii) a different stakeholder definition (inhabitants of the project area versus inclusion of local and national governments), and
(iii) the implications of severe budget constraints operating at the household level versus unconstrained cost calculations in van Beukering et al. (2003; see also discussion in Section 2).

Higher WTP may have been obtained if the proposed village development program had included micro-credit and agricultural extension schemes. Such a valuation frame may have alleviated existing budget and know-how constraints during the valuation interview. Also, the levels of the cost attributes did not exhaust the maximum WTP of the few 'rich' respondents. As an estimation of current local WTP for an important but narrowly defined ecosystem service in the face of severe poverty — 63% of project area inhabitants are not able to meet basic needs (van Rheenen et al., 2004), the seemingly low values reported here appear realistic.

In sum, the applicability of the CE instrument, the statistical significance of preferences for water availability, and most validity tests support the conclusion that the ecosystem service availability of irrigation water in the dry season was successfully valued. While most respondents sufficiently understood the ecosystem services to be valued, precautionary measures may be advisable to quantify the maximum impact of remaining unfamiliarity/comprehension problems.

5.2. *Data and modeling aspects*

Compared to the direct valuation of ecosystem functions, an ecosystem service approach places seemingly high demands on models and data needed to actually use WTP estimates to generate policy advice. Why demands may be hard to meet, can be explained from the modeling perspective introduced in Section 3.2 (Fig. 1). Because the ecosystem service approach does not confront respondents with technical descriptions of changes of ecosystem processes, structures and states ('ecosystem functions'), changes in these phenomena are not directly valuated. Therefore, the basic science models that represent these phenomena cannot be used directly to relate the valuated changes to changes in ecosystem functioning.

One example is the land use change in the Nopu catchment in Central Sulawesi. Detailed hydrological data exist how land use change affects variables of the hydrological cycle in this catchment (Kleinhans, 2004). While the underlying basic science model gives a precise description of hydrological ecosystem functioning, it cannot translate the land use changes into changes in the months with enough, barely enough and not enough water for irrigation purposes. This is no problem for Nopu where wet rice fields have been converted to cacao plots. It is a problem for the project area at large, however, where wet rice is a highly important subsistence crop. Still, village-specific engineering models that are capable of the necessary translations have not been completed, yet.

This is a typical situation for early phases of planning processes (Cerda et al. 2007). Here, economic valuation is not used for the appraisal of detailed project proposals. Instead, it is used for the identification and quantification of local preferences in order to improve the design of project or policy options. For informing early planning phases, choice experiments are particularly well-suited because (i) they do not have to assume linear preferences across the potential range of attribute level changes, and (ii) they generate trade-off information not only between a bundle of changes in environmental goods and income but also among the components of the bundle. These characteristics facilitate the meaningful application of stated preference results for the economic valuation of the final set of project or policy options.

6. Conclusion

Descriptive and normative interpretations of the ecosystem function and the ecosystem service concepts should be carefully distinguished. On this basis, the Central Sulawesi case study presents evidence for the applicability of an ecosystem service approach to the environmental valuation of potentially unfamiliar ecosystem functions. Three recommendations emerge from the case study to confront unfamiliarity in stated preference valuation:

(i) After extensive qualitative pre-studies, select demand-relevant ecosystem services for the description of the ecosystem function(s) to be valued,
(ii) use utmost care in the design and representation of the chosen set of ecosystem services, and
(iii) include variables that assess the potential impact of remaining unfamiliarity/comprehension problems.

To ensure the eventual policy relevance of valuation results, enough applied science and engineering expertise has to be available to translate changes in ecosystem functioning into meaningful ecosystem service descriptions, and to relate these changes to realistic policy or project interventions. Such translations remain a challenge for the entire discipline that only few studies – including revealed preference studies – have met (Heal et al., 2005, p. 154).

We conclude that the proposed ecosystem service approach can reduce the degree of unfamiliarity to a point where it does not pose a principal obstacle to the application of stated preference techniques. Because the number of ecosystem services that can be included in a survey study is limited, it will often only by possible to establish lower bounds of the total economic value of an ecosystem function. Still, it would constitute an unwarranted limitation of the diverse set of economic valuation techniques to exclude stated preference methods from the valuation of ecosystem functions on unfamiliarity grounds.

Acknowledgements

We thank Deutsche Forschungsgemeinschaft (DFG SFB 552) and BMBF (DEKLIM C) for financial support, and our Central Sulawesi respondents for their patience with our inquiries. Most appreciated advice and assistance was received from M. Bos, G. Burkard, D. Darusman, M. Fremerey, W. Lorenz, S. Schwarze, L. Sudawati, A.T. Tellu, M. Zeller, and our local enumerator team. Comments of two anonymous reviewers substantially helped improving the manuscript.

REFERENCES

Adamowicz, W., Boxall, P., Williams, M., Louviere, R., 1998. Stated preferences approaches to measuring passive use values: choice experiments and contingent valuation. American Journal of Agricultural Economics 80, 64–75.

Alcamo, J., et al., 2003. Millennium Ecosystem Assessment – Ecosystems and Human Well-Being: A Framework for Assessment. Island Press, Washington. 245 pp.

Arrow, K., Solow, R., Portney, P.R., Leamer, E.E., Radner, R., Schuman, H., 1993. Report of the NOAA Panel on Contingent Valuation. May 9, 2001. http://www.darrp.noaa.gov/library/pdf/cvblue.pdf.

Barkmann, J., Marggraf, R., 2003. Project A5 — Biological diversity of the rainforest margin as an economic good. Stability of Rainforest Margins, Fortsetzungsantrag 2003–2006, Georg-August-Universität Göttingen, pp. 229–263.

Barkmann, J., Marggraf, R., 2004. Ökologische Schäden durch Vernachlässigung des Vorsorgeprinzips im nachhaltigen Landschaftsmanagement — eine umweltökonomische Perspektive. In: Potthast, T. (Ed.), Ökologische Schäden. Peter Lang, Frankfurt/M., pp. 57–76.

Barkmann, J., Cerda, C., Marggraf, R., 2005a. Interdisziplinäre Analyse von Naturbildern: Notwendige Voraussetzung für die ökonomische Bewertung der natürlichen Umwelt Umweltpsychologie 9, 10–29.

Barkmann, J., Cerda, C., Marggraf, R., 2005b. Trading-off ecological insurance in an uncertain world: Economic preferences for species-diversity ensuring fundamental ecosystem functioning. Paper presented at DIVERSITAS 1st Open Science Conference, 9–12 November 2005, Oaxaca, Mexico.

Barkmann, J., Glenk, K., Handi, H., Sundawati, L., Witte, J.-P., Marggraf, R., 2007. Assessing economic preferences for biological diversity and ecosystem services at the Central Sulawesi rainforest margin — a choice experiment approach. In: Tscharntke, T., Leuschner, C., Zeller, M., Guhardja, E., Bidin, A. (Eds.), Stability of Tropical Rainforest Margins. Linking ecological, economic and social constraints of land use and conservation. Springer, Berlin, pp. 181–208.

Bateman, I.J., Carson, R.T., Day, D., Hanemann, M., Hanley, N., Hett, T., Lones-Lee, M., Loomes, G., Mourato, S., Özdemiroglu, E., Pearce, D.W., Sugden, R., Swanson, J., 2002. Economic Valuation with Stated Preference Techniques — A Manual. Edward Elgar, Cheltenham, UK. 458 pp.

Bennett, J., 2002. Non-Market Valuation Scoping Study — A report prepared for the Murray Darling Basin Commission, The Living Murray Project, September 2002. [http://www.thelivingmurray.mdbc.gov.au/__data/page/259/Non-market_valuation_scoping_report1.pdf].

Bennett, J., Blamey, R. (Eds.), 2001. The Choice Modelling Approach to Environmental Valuation. Edgar Elgar, Cheltenham (UK).

Birner, R., Mappatoba, M., 2002. Community-Based Agreements on Conservation in Central Sulawesi – A Coase Solution to Externalities or a Case of Empowered Deliberative Democracy? STORMA Discussion Paper Series, No. 3, Göttingen and Bogor.

Blamey, R., Rolfe, J., Bennett, J., Morrison, M., 1997. Environmental choice modeling: issues and qualitative insights. Research Report No. 4. Canberra, Australia: School of Economics and Management. The University of New South Wales.

Bockstael, N.E., Freeman, A.M., Kopp, R.J., Portney, P.R., Smith, V.K., 2000. On measuring economic values for nature. Environmental Science and Technology 34, 1384–1389.

Carpenter, S.R., Turner, M., 2000. Opening the black boxes: ecosystem science and economic valuation. Ecosystems 3, 1–3.

Carson, R.T., Flores, N.E., Mitchell, R.C., 1999. The theory and measurement of passive use value. In: Bateman, I.J., Willis, K.G. (Eds.), Valuing Environmental Preferences. Oxford University Press, Oxford, pp. 95–130.

Cerda, C., Diafas, I., Barkmann, J., Marggraf, R., 2007. WTP/WTA design strategies for choice experiments in early planning stages: experiences from Chile and Kenya. In: Meyerhoff, J., Lienhoff, N., Elsasser, P. (Eds.), Stated Preference Methods for Environmental Valuation: Applications from Austria and Germany. Metropolis Verlag, Marburg, pp. 139–173.

Costanza, R., Farber, S., 2002. Introduction to the special issue on the dynamics and value of ecosystem services: integrating economic and ecological perspectives. Ecological Economics 41, 367–373.

Costanza, R., d'Arge, R., de Groot, R.S., Farber, S., Grasso, M., Hannon, B., Naem, S., Limburg, K., Paruelo, J., O'Neill, R.V., Raskin, R., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Daily, G. (Ed.), 1997. Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC. 392 pp.

de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41, 393–408.

Diamond, P.A., Hausman, J.A., 1994. Contingent valuation: is some number better than no number? Journal of Economic Perspectives 8, 45–64.

Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecological Economics 41, 375–392.

Freeman, A.M., 1998. On valuing the services and functions of ecosystems. In: Freeman III, A.M. (Ed.), The economic approach to environmental policy: the selected essays of A. Myrick Freeman III. Edward Elgar, Cheltenham (UK).

Glenk, K., Barkmann, J., Marggraf, R., 2006a. Locally Perceived Values of Biological Diversity in Indonesia — A Choice Experiment Approach. [http://www.ucl.ac.uk/bioecon/8th_paper/Glenk.pdf].

Glenk, K., Barkmann, J., Schwarze, S., Zeller, M., Marggraf, R., 2006b. Differential Influence of Relative Poverty on Preferences for Ecosystem Services: Evidence from Rural Indonesia. Paper presented at the 26th Conference of the International Association of Agricultural Economists (IAAE), Queensland, Australia, August 12–18, 2006.

Heal, G.M., Barbier, E.B., Boyle, K.J., Covich, A.P., Gloss, S.P., Hershner, C.H., Hoehn, J.P., Pringle, C.M., Polasky, S., Segerson, K., Shrader-Frechette, K., Committee on Assessing and Valuing the

Services of Aquatic and Related Terrestrial Ecosystems, 2005. Valuing Ecosystem Services: Toward Better Environmental Decision-making. National Academies Press. 277 pp.
Hensher, D.A., Rose, J.M., Greene, W.H., 2005. Applied choice analysis: a primer. Cambridge University Press, New York. 717 pp.
Keil, A., Kleinhans, A., Schwarze, S., Birner, R., Gerold, G., Lipu, S., 2003. Forest conversion, water availability and water use in Central Sulawesi, Indonesia. Die Erde 134, 411–427.
Kleinhans, A., 2004. Einfluß von Landnutzung und pedohydrologischer Gebietsvarianz auf den Wasser-und Nährstoffumsatz in einem tropischen Einzugsgebiet (Zentralsulawesi). Dissertation, Georg-August-Universität Göttingen.
Kontoleon, A., Yabe, M., 2004. Assessing the impacts of alternative 'opt-out' formats in choice experiment studies. Journal of Agricultural Policy Research 5, 1–32.
Maertens, M., Zeller, M., Birner, R., 2004. Does technical progress in agriculture have a forest saving or a forest clearing effect? Theory and evidence from Central Sulawesi. In: Gerold, G., Fremerey, M., Guhardja, E. (Eds.), Land Use, Nature Conservation and the Stability of Rainforest Margins in Southeast Asia. Springer, Berlin, pp. 179–197.
Maertens, M., 2003. Economic modeling of agricultural land-use patterns in forest frontier areas: theory, empirical assessment and policy implications for Central Sulawesi, Indonesia. dissertation.de, Berlin.
Menzel, S., 2003. Der Beitrag der Protection Motivation Theory für die Zahlungsbereitschaftsäußerungen zur Erhaltung biologischer Vielfalt. Umweltpsychologie 7, 92–112.
Nunes, P.A.L.D., Bergh, J.C.J.M., 2001. Economic valuation of biodiversity: sense or nonsense? Ecological Economics 39, 203–222.
Pattanayak, S., Kramer, R.A., 2001. Pricing ecological services: willingness to pay for drought mitigation from watershed protection in eastern Indonesia. Water Resources Research 37, 771–778.
Rogers, R.W., Prentice-Dunn, S., 1997. Protection motivation theory. In: Gochman, D.S. (Ed.), Handbook of Health Behavior Research. Plenum, New York, pp. 113–132.
Schwarze, S., Zeller, M., 2005. Income diversification of rural households in Central Sulawesi, Indonesia. Quarterly Journal of International Agriculture 44, 61–74.
Steffan-Dewenter, I., Kessler, M., Barkmann, J., Bos, M., Buchori, D., Erasmi, S., Faust, H., Gerold, G., Glenk, K., Gradstein, S.R., Guhardja, E., Harteveld, M., Hertel, D., Höhn, P., Kappas, M., Köhler, S., Leuschner, C., Maertens, M., Marggraf, R., Migge-Kleian, S., Mogea, J., Pitopang, R., Schaefer, M., Schwarze, S., Sporn, S.G., Steingrebe, A., Tjitrosoedirdjo, S.S., Tjitrosoemito, S., Twele, A., Weber, R., Woltmann, L., Zeller, M., Tscharntke, T., 2007. Tradeoffs between income, biodiversity, and ecosystem functioning during tropical rainforest conversion and agroforestry intensification. Proceedings of the National Academy of Sciences of the United States of America (PNAS) 104, 4973–4978.
Sugden, R., 2005. Anomalies and stated preference techniques: a framework for a discussion of coping strategies Environmental and Resource Economics 32, 1–12.
Toman, M.A., 1996. Ecosystem valuation: an overview of issues and uncertainties. In: Simpson, R.D., Christensen Jr., N.L. (Eds.), Ecosystem Function & Human Activities. Chapman & Hall, New York, pp. 26–44.
van Beukering, P.J.H., Cesar, H.S.J., Janssen, M.A., 2003. Economic valuation of the Leuser National Park on Sumatra, Indonesia. Ecological Economics 44, 43–62.
van Rheenen, T., Elbel, C., Schwarze, S., Nuryartono, N., Zeller, M., Sanim, B., 2004. Encroachments on primary forest: are they really driven by despair? In: Gerold, G., Fremerey, M., Guhardja, E. (Eds.), Land Use, Nature Conservation and the Stability of Rainforest Margins in Southeast Asia. Springer, Berlin, pp. 199–213.
WBGU [Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen], 1999. Welt im Wandel – Umwelt und Ethik. Metropolis, Marburg. 149 pp.
Whigham, D.F., 1996. Ecosystem functions and ecosystem values. In: Simpson, R.D., Christensen Jr., N.L. (Eds.), Ecosystem Function & Human Activities. Chapman & Hall, New York, pp. 225–239.
Zeller, M., Wollni, M., Shaban, A., 2003. Evaluating the poverty outreach of development programs: empirical evidence from Indonesia and Mexico. Quarterly Journal of International Agriculture 42, 371–383.